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Enquiring into the roots of bioenergy
-
Epistemic uncertainty in life cycle assessments

Koldo Saez de Bikuña Salinas

PhD Thesis
March 2017

DTU Environment
Department of Environmental Engineering
Technical University of Denmark

Enquiring into the roots of bioenergy – epistemic uncertainty in life cycle assessments

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The synopsis part of this thesis is available as a pdf-file for download from the DTU research database ORBIT: http://www.orbit.dtu.dk.

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Preface

The thesis is organized in two parts: the first part puts into context the findings of the PhD in an introductive review; the second part consists of the papers listed below. These will be referred to in the text by their paper number written with Roman numerals (Papers **I-III**).

I. Environmental performance of gasified willow from different lands including land-use changes. GCB Bioenergy 2016. doi: 10.1111/gcbb.12378.

II. Framing the time horizon conundrum in biofuel assessments: a comparison of land use change accounting methods. Submitted to Environ. Sci. Technol.

III. Environmental flows and stocks: reconciling modelling approaches with land-use references in land use impact assessments. To be submitted to Int. J Life Cycle Assess.

In this online version of the thesis, **papers I-III** are not included but can be obtained from electronic article databases, e.g. via www.orbit.dtu.dk or on request from:

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Acknowledgements

An honest expression of intimate emotions is always an act of soul nudity. That is why this non-technical part may be the most challenging to write of all. Those who know me know that I like languages – that human vehicle to transport thoughts and emotions from one mind to another enticed me, always an entertaining exploration of the different human universes of expression. Since the many people I feel gratitude for speak different languages (and not all of them understand English), this shall be a multi-lingual acknowledgments chapter.

Jeg skal takke først Claus Beier, who chose and trusted me for carrying out this complex white-cheque research job and DTU Kemiteknik for financing it without constraints. Et stor tak til alle ECO venner og særlig min vejleder Andreas Ibrom, for his Deutsch discipline to keep a passionate Iberian on track and (kind of) in time. The many discussions, not always related to the PhD, helped me keep going. I particularly appreciate his courage and confidence when he allowed me becoming part-time without "PhD-relevant" or "normal" reasons and against his initial will (and probably contrary to his intuition too). Tusind tak også til Michael og de fantastisk QSA crew too, for hosting me during some months in his dynamic, engaging and inspiring research team. Sidst, men ikke mindst, jeg skal selvfølgelige takke DTU Miljø for welcoming these countryside researchers (once upon a time in Risø…) and hosting our dear ECO group (now AIR section); it was surprisingly easy to fit in and you made all of us feel like being at a new home.

I would surely like to thank my big International Family of Copenhagen as well: all my south-European PIGS friends, the climber-freaks, i soliti (s)kazzoni, my ex-flatmates and all the movies4masses people: the Mordor periods were brighter with you around and the summer periods were almost like being in the far South – except for the weather. And the mountains. And the sea. And the food. And the open-air parties. And the prices... In any case, thanks to all of you! ☺

Επειδή αυτή η Οδύσσεια δεν θα ήταν δυνατή χωρίς εσάς : ο Κολντάκος θα είχε ναυαγήσει μεταξύ της Καλυψώς και των λωττοφάγων, μη φτάνοντας ποτέ στην Ιθάκη. Μαζί μπορούμε να φτάσουν Ιθάκη, ή απλά ένα άλλο νησί ;)

Y para terminar, recordar a mi familia, la de verdad, txikia ta baita haundia ere. El camino fue largo y pedregoso, como el de aquellos hobbits hasta llegar a la montaña de Sauron para destruir el anillo único. Algunos perdimos

por el camino, otros se quedaron a medio camino para luego volver, como los gatos siete veces (¡te quedan cinco!)... Es bonito sentirse chiquitito en una manada grande. Además, tengo la suerte de poder refugiarme a voluntad en mi segunda lengua madre, el euskera, que tanto estimo, para poder declararme abiertamente pero con cierta intimidad, a quien más estimo:

Tesi hau zuei eskeinita doa, guraso maitagarri ta maitaleok, hasieratik amaierararte: jaso izan dudan guztia bueltatu ezinean, betidanik ta betirako zordun izango den zuen seme kuttun ero hone partez, ta bihotz-bihotzez,

Eskerrik asko!!!
¡¡Gracias de corazón!!
Ευχαριστώ;;
Grazie mille!!
Tusind tak!!
Merci beaucoup!!
Thanks a lot!!

Koldo

Summary

The research for this Thesis was originally framed around the "sustainability assessment of full chain bioenergy". However, it is known for some years that the critical impacts of dedicated bioenergy relate to induced land use changes (LUC). Their criticality derives from their potential to dominate environmental impacts from a life-cycle perspective and from the uncertainty that accompanies them. On the other hand, continued land use may be a concern for soil's long-term sustainability (understood as fertility), which has recently received attention in environmental life-cycle assessments (LCA) under the respective life-cycle initiative of the UNEP-SETAC. The Thesis thus focused on these two aspects of sustainability of bioenergy. The overall aim was to disentangle the epistemic uncertainties related to land use impact assessments in order to provide science based decision-support for environmentally sustainable land use management and policy-making, especially relevant for land-demanding or dedicated bioenergy deployment.

Paper **I** took a Danish willow plantation for cogeneration of heat and power (CHP) through gasification and framed the research around the key land-use reference assumptions. For this, the LCA was structured around three basic land scenarios: marginal abandoned land, marginal extensive grassland and arable land. For each scenario, different LUC models were developed which represent the different impacts induced from the occupation of land for energy cropping. Despite being the most productive, occupying arable land proved to have the largest impacts due to indirect LUC. Gasification willow from marginal abandoned land had also significant impacts from preventing natural regeneration, but it showed a significantly better environmental performance (even under the considered uncertainties) than CHP from natural gas. The implementation of such bioenergy systems on abandoned lands would be thus justified as long as they substitute fossil-fuel based CHP.

In Paper **II**, the key assumptions related to time horizons in LCA of bioenergy systems were analysed and crucial definitions for them were proposed, as well as generic recommendations regarding them. Similarly, the effect of different modelling approaches in LUC emission accounting was studied by the application of several methods to four biofuel case studies. As a result, dynamic land-use baseline methods were rejected for LUC accounting while top-down LUC models showed to be a more solid alternative to economic iLUC models for regulation and footprinting purposes. After considering the studied epistemic uncertainties and based on the key conservative assump-

tions taken, it was concluded that land-demanding biofuels have larger global warming impacts than the respective fossil fuels they replace unless planted on abandoned lands.

With Papers **I-II**, the selection of the land-use references and time horizons involved in LCA of biofuels was demonstrated to be crucial for the characterization of the resulting environmental impacts. On top of that, different LCA modelling approaches exist with different virtues and applications, which logically articulate different sets of other key assumptions. Therefore, three land-use reference frameworks were proposed in Paper **III** to enable value-consistent land use impact assessments. Based on previous findings and recommendations, new methodological modifications to the existing UNEP-SETAC framework were suggested. The proposed modifications were articulated by discriminating among different long-term impacts from land use and by classifying different ecosystem services provided by land as environmental stocks or flows. These modifications reorient the land use impact assessments to impacts *during occupation* and suggest dealing with permanent impacts separately. In the proposed new methodology, dynamic land-use references are suggested for assessing occupation impacts on abandoned lands (relevant for consequential LCA) while static references are suggested for generic occupation impacts (in any LCA). Static references, understood as the precedent vegetation cover in equilibrium, are also suggested for every transformation impact assessed with any LCA modelling approach.

Last but not least, a hybrid LCA (HLCA) framework was also proposed as an alternative to existing attributional LCA which facilitates both *absolute and relative* sustainability assessments. Unknown or indirect LUC can be included with top-down LUC models (LUC_{global} factor for world-average greenhouse gas (GHG) emissions or $LUC_{GHGprotocol}$ factors for country-average, crop-specific GHG emissions). In order to enable absolute land use impact assessments, the use of substitution is not allowed in the HLCA framework and an area based functional unit (FU) should be chosen. For this, land use impacts can be linked to planetary and regional ecosystem boundaries through normalization references (taken as carrying capacity thresholds). Environmental footprinting of products from land-use systems with co-products can be carried out by choosing product-based FU, but absolute land use impact assessments would involve then value-laden allocation choices. Value-free absolute impact assessments can still be carried out with area-based FU and by adding function-equivalent synthetic products to the other system(s), which allow system (rather than product) comparability.

Dansk sammenfatning

Den oprindelige baggrund for denne afhandling var ”vurdering af bæredygtighed af bioenergi fra *vugge til grav*”. I nogle år har det dog være kendt at betydelige effekter af bioenergi er relateret til ændringer i arealanvendelse (LUC). Betydningen af denne effekt skyldes disse ændringers potentiale til at dominere miljøpåvirkningerne fra et livscyklus perspektiv og fra den usikkerhed der følger dem. På den anden side kan vedvarende drift af arealer være afgørende for deres bæredygtighed på langt sigt (med hensyn til frodighed); dette har i den seneste tid fået øget opmærksomhed i miljømæssige livscyklusvurdering (LCA) under initiativ af UNEP-SETAC. Denne afhandling fokuserer på disse to aspekter af bæredygtighed af bioenergi. Hovedformålet var at tilvejebringe videnskabeligt funderet beslutnings-støtte for miljømæssig bæredygtig arealanvendelse og politiske retningslinjer, specielt relevante for arealkrævende dedikeret bioenergiproduktion.

Artikel I omhandler en dansk energipil bevoksning dyrket med henblik på kraftvarme produktion (CHP) ved forgasning af biomassen, hvor artiklen fokuser på nøglereferencer for arealanvendelse. Med dette formål, er livscyklusvurderingen struktureret omkring tre arealanvendelses-scenarier: marginaljord uden drift, marginaljord med vedvarende græs og landbrugsjord. For hvert scenarie blev der konstrueret forskellige LUC modeller, der re-præsenterer de forskellige påvirkninger fra den ændrede arealanvendelse ved dyrkning af bioenergiafgrøder. På trods af at være det mest produktive, viste dyrkning af bioenergiafgrøder på landbrugsjord at give den største påvirkning på grund af indirekte LUC. Forgasning af pileflis fra marginaljord resulterede også i en betydelig påvirkning ved at forhindre naturlig genvoksning, men dog med en betydelig mindre miljøpåvirkning end ved kraftvarme produktion fra naturgas. Høst af energiafgrøder fra marginal-jorde er således fordelagtige så længe de erstatter fossilt baseret kraftvarmeproduktion.

I **Artikel II** analyseres nøgle-forudsætningerne for LCA med hensyn til tidshorisonten for dyrkningssystemer for bioenergi, og afgørende definitioner og befalinger opstilles for tidshorisonten. Effekten af forskellige tilgange til modelling af drivhusgas (GHG) emissioner som følge af ændring i arealanvendelse blev undersøgt v.hj.a. forskellige metoder anvendt på fire biobrændstof case studier. Resultatet af denne analyse er, at dynamiske reference arealanvendelses metoder blev forkastet for LUC opgørelse, mens ”top-down” LUC modeller viste sig at være et godt alternativ til økonomisk baserede indirekt LUC modeller med henblik på regulering og økologisk fodaftryk. Efter analyse af de videnskabeligt baserede usikkerheder og med basis i de konservative forudsætninger, blev det konkluderet at arealkrævende bioenergiafgrøder har større global opvarmningspåvirkning end de respektive fossil-baserede energiformer, som de skal erstatte, med mindre de dyrkes på marginaljorde.

I artiklerne I og II blev det påvist, at valget af reference-arealanvendelse og tidshorisont for LCA af bioenergiafgrøder er afgørende for karakteriseringen af den resulterende miljøpåvirkning. Derudover eksisterer der forskellige LCA modelleringsmetoder med forskellige egenskaber og anvendelsesområder, der logisk fremhæver forskellige nøgle forudsætninger. Derfor blev tre forskellige referencerammer for ændret arealanvendelse foreslået i **Artikel III** med henblik på værdi-konsistent bedømmelse af ændret arealanvendelse. Med udgangspunkt i tidligere undersøgelser og anbefalinger foreslås ændrin-ger til den eksisterende UNEP-SETAC metode. De foreslåede ændringer er karakteriseret ved at diskriminere mellem forskellige lang-tids effekter fra arealanvendelse og ved at klassificere forskellig økosystem-tjenester der ydes af jorden som miljømæssige ”stocks” eller ”flows”. Disse ændringer flytter fokus for vurderingen af påvirkningen til den aktuelle periode for dyrkningen af bioenergiafgrøden, og foreslår at vedvarende ændringer behandles separat. I den nye metode foreslås dynamiske referencer for arealanvendelse for be-dømmelse af effekten på marginaljorde (relevant for ”consequential” LCA), mens statiske referencer foreslås for påvirkninger under almindelig dyrkning (i alle LCA). Statiske referencer, forstået som det oprindelige vegetations-dække i ligevægt, foreslås for enhver effekt af ændringer i LCA modellering.

Sidst, men ikke mindst, foreslås en hybrid LCA (HLCA) ramme som et alter-nativ til den eksisterende traditionelle ”attributional” LCA, som kan håndtere både *absolut og relativ* bæredygtigheds vurdering. Ukendt eller indirekte LUC kan inkluderes igennem top-down LUC modeller (LUC_{global} faktorer for globale gennemsnit for GHG emissioner eller med $LUC_{GHGprotocol}$ faktorer for lande gennemsnit af afgrøde-specifikke GHG emissioner). Med henblik på at opstille absolutte bedømmelser af effekten af arealanvendelse er brugen af substitution ikke tilladt i HLCA og en arealbaseret funktionel enhed (FU) skal anvendes. For at opnå dette kan effekter af arealanvendelse kobles til globale og regionale økosysten grænser gennem normaliserende referencer (beregnet som tærskler for bæredygtigheds-kapacitet). Produkters miljømæs-sig fodaftryk fra arealanvendelse med flere produkter kan udføres ved at væl-ge produktbaserede FU, men absolut vurdering af miljøpåvirkning fra areal-anvendelse vil indebære værdi-baserede fordeling (dvs. ”allocation”). Værdifrie absolutte vurderinger af effekt kan stadig udføres med areal-baserede FU og ved at inkludere funktions-ækvivalente syntetiske produkter til de andre systemer, hvilket gør sammenligning af systemer (i stedet af produkter) mulig.

Table of contents

Abbreviations

LCA	Life Cycle Assessment
CLCA	Consequential Life Cycle Assessment
ALCA	Attributional Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
FU	Functional Unit
CF	Characterization Factor
LUC	Land Use Change
iLUC	Indirect Land Use Changes
dLUC	Direct Land Use Changes
OI	Occupation Impacts
TI	Transformation Impacts
DRI	Delayed Relaxation Impacts
GW	Global Warming
GWP	Global Warming Potential
IPCC	Intergovernmental Panel on Climate Change
UNEP	United Nations Environment Program
SETAC	Society of Environmental Toxicology And Chemistry

We do not owe the freshness of the air or the sparkle of the water. How can you buy them from us?

Massasoit, American native chief

Only when the last tree has been cut, the last river has been poisoned and the last fish has been caught will white man realize that humans cannot eat money.

American native Cree people

~ ~ ~

Scientific knowledge is to know through demonstration. Certain knowledge necessarily departs from true, immediate, non-demonstrable premises, like the commensurability of the diameter. Premises are the primary causes of conclusions.

Aristotle, *Posterior analytics*.

The necessity to decide is always greater than the level of cognition.

Kant

The habit of analytical thought is fatal for the intuitions of the integral, holistic thought.

Aldous Huxley, *The perennial philosophy*.

1 Introduction

Over the last decades, a gradual shift from fossil fuels to renewable energy sources has ensued in many industrialised countries since the Kyoto Protocol (UNFCCC 1998), further advanced with the Paris Agreement of the COP21 (UNFCCC 2015). Bioenergy is deemed to be part of the future energy mix and one of the inevitable alternatives to substitute fossil fuels (Grassl et al. 2003; Bauen et al. 2009). Even though the available additional bioenergy potential is in dispute (Beringer et al. 2011; Smith et al. 2012; Slade et al. 2014), it is unquestionable that biomass will be a necessary feedstock in the future for the production of biomaterials. Moreover, biomass can also be converted to storable energy carriers. Solid biofuels can compensate for the fluctuations of other intermittent renewable energy sources and power transmission limitations of the grid, while liquid biofuels can substitute fossil fuels with difficult alternatives like aviation or remote applications.

The controversy of land-demanding bioenergy does not only regard its potential, but also the multiple environmental impacts on land that may arise from scaling-up their production (Searchinger and Ralph 2015). These include increasing pressure on water (Gerbens-Leenes et al. 2009), biodiversity (Koh 2007; Dale et al. 2010; Dauber et al. 2010) and climate (Fargione et al. 2008; Searchinger et al. 2008; Lapola et al. 2010; Schulze et al. 2012). Given the enormous demand on productive land it may trigger if scaled-up (Foley et al. 2005), bioenergy cropping can also spark acute social conflicts (Homer-dixon 1991; Homer-dixon 1994) and severe problems among vulnerable population in countries with poor governance derived from large-scale farmland investment and land grabbing phenomena (Cotula et al. 2008; Borras et al. 2011; De Schutter 2011; Deininger 2011; GRID-Arendal 2013; Baka 2014).

The current scientific challenge in land-demanding bioenergy research relates though to the quantification of the indirect effects and land use change (LUC) emissions. Reported greenhouse gas (GHG) emissions from land use and LUC vary drastically with the assumed time horizons (Kløverpris and Mueller 2012; Valin et al. 2015) and land-use references (Soimakallio et al. 2015) in life cycle assessments (LCA). To this, other assumptions needed in agro-economic models have led to a great result variability of indirect effects in literature (Plevin et al. 2010; Broch et al. 2013; Smith et al. 2014). The large variance of reported indirect LUC (iLUC) emission factors has resulted in great uncertainty in the decision support for biofuel policy-making (Finkbeiner 2014; European Commission 2015). This variance is an indicator of the knowledge gap that needs to be addressed (Muñoz et al. 2014) and suggests that, besides parametric uncertainties (Plevin et al. 2015), epistemic uncertainties are also involved in iLUC emissions estimations related to the many assumptions behind the agro-economic models. Consequently, assessing the plausibility of critical assumptions is deemed a prerequisite for the advancement of LCA as an effective decision-support tool for land use management and biofuel policy-making.

On the other hand, LCA has been lacking a specific methodology to address and frame the multiple impacts that arise from land use and LUC until recently (Koellner et al. 2013). The methodology proposed by the UNEP-SETAC framework arrived later than other life cycle impact assessment (LCIA) methods due to the intricate and complex interconnections of the multiple mechanisms governing terrestrial ecosystems (Millennium Ecosystem Assessment 2005). The methodology is based on previous work that addressed the hitherto unattended soil quality in LCA where soil organic matter (SOM) was proposed as a single proxy indicator (Milà i Canals 2003; Milà i Canals et al. 2007). The method was further developed to cover several soil quality aspects (with five midpoint indicators) (Beck et al. 2011). This was in turn expanded by the UNEP-SETAC life cycle initiative to build a more complete framework that covers land use impacts on soil quality (Saad et al. 2011), soil productivity (Brandão and Canals 2012), climate (Müller-Wenk and Brandão 2010) and biodiversity (Baan et al. 2012).

1.1 Research problem and research objectives

To the inherent uncertainty of economic iLUC models, discrepancies around critical time horizon and land-use reference assumptions in LCA add up to the biofuel conundrum. Thus a need for a clearer understanding of the relationships between induced LUC emissions, land use impact assessment results in biofuel LCA and the key assumptions behind them has been identified. This is seen as a prerequisite to develop further the existing land use impact assessment methodology proposed by UNEP-SETAC.

The main aim of this PhD Thesis is hence to provide science based decision-support for environmentally sustainable land use management and policy-making, particularly those concerning bioenergy deployment. To achieve this, the following objectives were pursued:

- Assess the effect of land-use reference choices in LCA results of bioenergy systems.

- Identify the different time horizon assumptions involved in LCA of bioenergy systems.
- Assess the effect of modelling approaches in LUC emission accounting results of land-demanding bioenergy systems.
- Develop a framework to consistently guide practitioners in the selection of land-use references in land use impact assessments in LCA.
- Study the feasibility of a new approach that reduces the epistemic uncertainties related to economic iLUC models.
- Further develop the land use impact assessment framework and methodology of the UNEP-SETAC with learned research outcomes from the Thesis.

1.2 Content of the PhD Thesis

In order to meet the stated research objectives, this Thesis is structured as follows:

- Chapter 2 describes the research design of the Thesis. Here the general research method is succinctly explained and the involved worldviews and underlying theories are presented.
- Chapter 3 presents a systematic analysis of land-use reference effects in the LCA results of a willow bioenergy case study (Paper **I**).
- Chapter 4 presents an analysis of the assumptions related to time horizons in LCA of bioenergy systems and suggests definitions for them (Papers **II-III**).
- Chapter 5 shows the effect of different modelling approaches in LUC emission accounting, illustrated by the application of several methods to four biofuel case studies (Paper **II**).
- Chapter 6 presents the modification proposals for the current land use impact assessment methodology of the UNEP-SETAC framework in LCA. (Paper **III**).
- Chapter 7 discusses the limitations of current LCA and the advantages and challenges of future absolute sustainability assessments with LCA.
- Chapter 8 concludes with the main research outputs and recommendations of this PhD Thesis.
- Chapter 9 outlines the perspectives for future research in LUC modelling and land use impact assessments with LCA.

2 Research design

Evidently, we need to arrive to premises through induction. It is intuition that apprehends the primary premises, since these cannot be demonstrated from anything else. Intuition is thus the source of scientific knowledge.

Aristotle, *Posterior analytics*.

The general research method or epistemological approach taken to generate new knowledge is the inductive method. The Thesis departs from specific bioenergy study cases (Chapters 3 and 5) to assess the effects of different key assumptions in LCA results, inducing more general principles that apply to any land-demanding product system (Chapters 4 and 6). These, in turn, enable the reformulation and reframing of the existing land use impact assessment methodology and framework in LCA (Chapter 6).

Similarly, the philosophical worldview of this study and the way it has shaped the approach to study the interaction between modelled social systems and the environment is here presented. The author has deemed necessary this clarification, given that the current PhD Thesis deals with research on *sustainability*. And sustainability researchers are doomed to take a stand. As the reader may know, sustainability is a broad multidisciplinary science that sways between environmental sciences and social sciences (including economics). From an epistemological viewpoint, the former scientific discipline is more objective than the latter for it deals with *objects* of study (which can be described through pure bio-physico-chemical models), while the latter deals with *subjects* of study that require additional (and sometimes problematic) assumptions (e.g. a presupposed rational behaviour of individuals) or new agent-based models (which incorporate the inherent complexity and randomness of studied autonomous agents).

Since LCA, not agent-based modelling, has been chosen to provide science-based decision-support for environmentally sustainable land use management, this Thesis has focused on the study of epistemic uncertainties derived from *critical assumptions* in land use impact assessments. Critical assumptions are those which are both *sensitive* (i.e. which have a high influence on the results) and *uncertain* (i.e. for which current scientific knowledge cannot discern as more plausible under certain conditions).

2.1 Philosophical worldview

The current Thesis lies within the *post-positivist* epistemological school, which accepts that underlying theories, values and background of the researcher can influence what is observed and modelled (Popper 1959). This school suggests that, despite objectivity being pursued, reality can only be known imperfectly and probabilistically due to the effect of biases. Recognizing the inescapable stance of the modeller, multiple view-points are hence incorporated into the analysis and problem solving process. Likewise, the author acknowledges that, in the light of indisputable and increasing evidence of climate change and anthropogenic-related environmental problems, science is currently facing a whole paradigm shift or change of philosophical worldview (Kuhn 1962). The resulting research-front of sustainability science is thus embedded in the paradigm of *post-normal science* (Funtowicz and Ravetz 1992), which refers to the precautionary stance of scientists whose research aims at decision-support in the light of high uncertainty (see Figure 1).

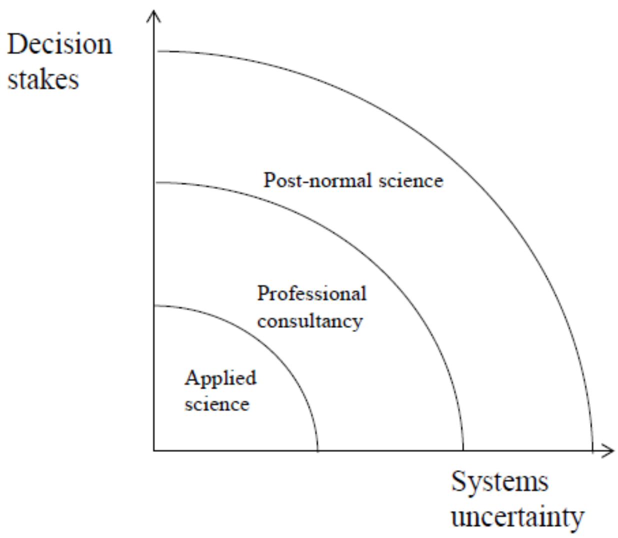

Figure 1. Diagram of post-normal science adapted from Funtowicz and Ravetz 1992. Increasing decision stakes and systems uncertainties entail new problem solving strategies.

2.1.1 Theoretical framework: the underlying theories

After introducing Cultural Theory into LCIA modelling, it was understood that the selection of one perspective from the “value-sphere” determines the perception, description and modelling of the interactions between the eco-sphere and the techno-sphere (Hofstetter 1998). The current Thesis does not aim at rebuilding LCA on the basis of the five social archetypes suggested by Hofstetter, but rather at bringing it forward by recognizing the systematic bias that practitioners and modellers introduce in the different stages of LCA which affect the results (Thomassen et al. 2008).

Last but not least, the author recognizes both the value and the limitations of purely *relative* sustainability assessments of standard LCA. They facilitate a much needed guidance and scientific support for the progressive advancement of the design, management and policy-making of environmentally friendli*er* products. But we also know this is not good enough (Bjørn et al. 2015). A truly integral approach must incorporate concerning tipping-points of ecosystems (Rockström et al. 2009; Barnosky et al. 2012) to enable *absolute* sustainability assessments as well. Without digging into the underlying theories, the author acknowledges here the economic schools behind these two general approaches to sustainability: i.e. *environmental* (Perman et al. 2011) and *ecological* (Odum 1973; Daly 1974) economics. They are distinguished by their approach towards sustainability, defined by a “weak” or a “strong” criterion, respectively. These criteria are more thoroughly presented and discussed in Chapter 7 of the Thesis.

3 The effect of land-use references in LCA of biofuels

The type of land assumed to be taken into cultivation is known to be a critical factor in iLUC results (Broch et al. 2013). The first study case thus aimed at analysing the consequences of occupying different land types in Denmark for energy cropping, including market-mediated effects like iLUC (Paper **I**). Therefore, a consequential LCA (CLCA) modelling approach was chosen for the purpose. A real Danish short-rotation coppice (SRC) willow plantation with a low-input management regime of 20 years of duration was taken. The crop is harvested every three years and the woodchips are used for cogeneration of heat and power (CHP) through gasification in a nearby 1.5 MW_{th} input plant. Primary data for most farming and energy conversion processes was taken, while for the rest of the background processes (e.g. fertilizer production) the Ecoinvent LCA database was used (Ecoinvent 2014).

The study was structured around three basic land scenarios that could be used in Denmark for energy cropping: common arable land, marginal extensive grassland and a hypothetical marginal abandoned farmland. The occupation of each land type for energy cropping triggers soil organic carbon (SOC) changes, but also other LUC effects: iLUC (for the first two scenarios where the production of food or fodder are assumed to be displaced) and direct LUC (dLUC, for the abandoned land scenario). For the first two scenarios, different iLUC models were developed: $iLUC_{food}$ (for energy cropping on arable land), $iLUC_{feed}$ (for energy cropping on marginal grassland). For the abandoned land scenario, $dLUC_{aban}$ (simply referred to as dLUC in Paper **I**) was developed as the foregone C sequestration from the prevention of natural regeneration. While dLUC on the arable land and grassland scenarios include only SOC changes (from a C flow model and a meta-analysis, respectively), $dLUC_{aban}$ considers SOC and biotic carbon (BioC) changes (see Table 1).

3.1 Applied land-use references in the three basic land scenarios

In this CLCA, the business-as-usual (BAU) land management concept defines the different land-use references that apply to each land scenario (see Table 1). The resulting land use impacts are characterized by the applied LUC model and the respective reference. The reference is in turn defined by the type of land (directly or indirectly) occupied.

For the dLUC models of the arable land and the marginal grassland scenarios, the references are *intensive wheat production* (REF1) and *extensive land use management* (REF2), respectively. To calculate occupation impacts (OI) on abandoned lands, the reference is *dynamic* (REF3, used in the $dLUC_{aban}$ and $iLUC_{food}$ models) since nature would gradually take over the land in the absence of the studied system. However, the resulting OI are named as delayed relaxation impacts (DRI) for land use situations with REF3. For the case of virgin lands without human influence, a *static* reference is taken to calculate transformation impacts (TI) since the disturbed ecosystem would have remained in its natural equilibrium without the studied system (REF4). For the case of arable land affected by the indirect intensification effect, the reference is local semi-intensive land management (REF5). Therefore, OI apply to land use situations with REF1, REF2 and REF5 (see Table 1).

Table 1. Framework showing the applied land-use references in each LUC model of the three basic land scenarios. Adapted from Paper **I.**

ARABLE LAND		MARGINAL GRASSLAND		MARGINAL ABANDONED
dLUC C flow model (C-TOOL)	**$iLUC_{food}$** Hybrid model (economic + top-down)	**dLUC** Meta-analysis	**$iLUC_{feed}$** Top-down model	**$dLUC_{aban}$** Linear model (SOC + BioC)
REF1: Intensive wheat production (OI)	REF3: Natural regeneration (DRI) REF4: Natural vegetation cover (TI) REF5: Local arable management (OI)	REF2: Extensive pasture management (OI)	REF4: Natural vegetation cover (TI) REF5: Local arable management (OI)	REF3: Natural regeneration (DRI)

3.1.1 Delayed relaxation impacts on marginal abandoned land

To assess land use impacts on abandoned lands, a dynamic land-use reference (REF3) is taken. This is used to calculate $dLUC_{aban}$ and part of the $iLUC_{food}$ model. It implies calculating C losses relative to a natural regeneration over 20 years. The process was assumed linear taking the total BioC of a managed beech forest as the 100 year endpoint (Wu et al. 2013), which yielded a relative BioC loss of 4116 kg CO_2e ha_{occup}^{-1} yr^{-1}. Adding the relative SOC loss, the resulting $dLUC_{aban}$ turned to be 4231 kg CO_2e ha_{occup}^{-1} yr^{-1}.

3.1.2 Top-down iLUC model for the marginal grassland: $iLUC_{feed}$

A top-down approach was followed to determine the $iLUC_{feed}$ emission factor. This is composed of a global land transformation ($iLUC_{feed\_trans}$) and an intensification ($iLUC_{feed\_int}$) factor. Taking FAO statistics for global deforestation and average agricultural expansion-intensification shares of global food production (37% and 63%, respectively) (Tonini et al. 2015), an annual global demand of 27.7 Mha_{dem}-eq yr^{-1} of new productive land was estimated. This way, the $iLUC_{feed\_trans}$ factor was calculated combining country-specific FAO deforestation data with IPCC data for biome C stocks of the affected regions and other non-CO_2 GHG emission factors from LUC. To calculate these transformations, a REF4 was used and the resulting emissions were amortized over the occupation period.

On the other hand, the $iLUC_{feed\_int}$ factor was calculated as the annual world-average change in N-fertilizer production from FAO statistics. For these intensification impacts, a REF5 was used (assuming linearity between increased intensification and OI). Nonetheless, it was assumed that the displaced grass is substituted with a commercial feed mix of soybean meal and maize grain of an equivalent nutritive value (Tonini et al. 2015). This means in practice that a ratio of demanded commercial feedstock area per occupied grassland area (0.93 ha_{dem} ha_{occup}^{-1}) is applied to the shown $iLUC_{feed}$ factors, which is based on the nutritive value of the grass displaced. The shown $iLUC_{feed\_trans}$ factor thus represents the average GHG emissions from induced deforestation per additional area demanded. The $iLUC_{feed\_int}$ factor represents the average GHG emissions from the induced production and use of N-fertilizer per additional area demanded.

3.1.3 Hybrid iLUC model for the arable land: $iLUC_{food}$

For the hybrid $iLUC_{food}$ model, the area expansion results from a general equilibrium economic model simulation (Kløverpris 2008) were combined with the previous top-down approach to include intensification effects. The novel part of this hybrid model is that includes the three main indirect effects that are triggered from occupying arable land for energy cropping (which displaces the still demanded wheat production). The included indirect effects are: land transformation (as new agricultural area expansion onto virgin ecosystems), delayed relaxation (as the share of the area expansion onto marginal lands that had been abandoned) and intensification (as increased application of synthetic N-fertilizers in existing cropland).

On one hand, the area expansion results from the economic model were divided among transformation or delayed relaxation impacts. Transformation of virgin ecosystems was considered in the world regions with historical deforestation trends and delayed relaxation in the areas where historical land abandonment had been identified (Ramankutty et al. 2008; Kløverpris 2009). Therefore, regionalized characterization factors (CF) per area transformed or re-occupied (CF_{trans} and CF_{DR}, respectively) were produced in Mg C ha_{exp}^{-1} (reported in Table 3 in Paper **I**). The CF_{trans} were calculated respect to REF4, i.e. as the locally induced BioC and SOC losses, amortized over the occupation period while CF_{DR} were calculated respect to REF3, i.e. as the foregone C sequestration from postponing natural relaxation for 20 years. The resulting CF_{trans} and CF_{DR} were then multiplied by the induced regional area expansion per tonne wheat demanded (presented as A_{iLUC} in Table 3 in Paper **I**) and the average Danish wheat yield to obtain the area-related GHG emissions per area demanded (reported in Mg CO_2eq ha_{dem}^{-1}). The intensification share of the global wheat production response used in the economic model was finally applied to the previous top-down intensification factor (30% instead of the global average of 63% as in $iLUC_{feed}$) to come up with the $iLUC_{food\_int}$ factor (see Table 2).

Table 2. Results for developed LUC models: linear ($dLUC_{aban}$), hybrid ($iLUC_{food}$) and top-down ($iLUC_{feed}$). Shown GHG emissions from transformation have been amortized over the occupation period (20 years). Adapted from Paper **I.**

Land scenario	LUC factors	Land-use reference	Area ratio* (A_{iLUC}/A_{occup})	GHG emissions (kg CO_2eq ha_{occup}^{-1} yr^{-1})	Share (%)
Abandoned land	**$dLUC_{aban}$**	REF3	0	**4231**	100
Arable land	$iLUC_{food\_trans}$	REF4	0.881	11715	82
	$iLUC_{food\_DR}$	REF3	0.319	1560	11
	$iLUC_{food\_int}$	REF5	0.515 ǂ	962	7
	$iLUC_{food}$	**-**	**1.715 ǂ**	**14236**	**100**
Marginal grassland	$iLUC_{feed\_trans}$	REF4	0.345	7436	80
	$iLUC_{feed\_int}$	REF5	0.588 ǂ	1885	20
	$iLUC_{feed}$	**-**	**0.933 ǂ**	**9322**	**100**

* A_{occup} is presented as A_{SRC} in Paper **I**. ǂ Represent area-equivalents.

3.2 Key LCA results

Global warming (GW) and acidification impacts of gasification willow are lower and significantly lower than natural gas and coal, respectively (see Table 3). The toxicity and eutrophication impact potentials for gasification willow are slightly lower and slightly higher, respectively. When GHG emissions from induced LUC effects ($iLUC_{food}$, $iLUC_{feed}$, $dLUC_{aban}$) are considered, these play a crucial role in the GW impact of the willow-gasification bioenergy system. Land transformation dominates the impacts from considered LUC effects (see Table 2). Land transformation is also the main driver of biodiversity impacts since induced deforestation occurs mainly on biodiversity-rich tropical ecosystems with endemic and endangered species. Foregone C sequestration and potential loss of regional biodiversity from energy cropping on abandoned lands can be seen as a trade-off for iLUC-free (thus global biodiversity friendly) bioenergy.

Table 3. LCA results (per *net* energy output) for the four selected impact categories. Note that willow bioenergy results are credited with avoided natural gas. Taken from Paper **I.**

Energy system	GW_{100} (g CO_2eq MJ^{-1})	Eutrophication (mg PO_4^{3-}eq MJ^{-1})	Acidification (mg SO_2eq MJ^{-1})	Toxicity (g DCBeq MJ^{-1})
CHP willow (arable)	0.8	23.2	29.1	409.6
CHP willow (marginal grassland)	-10.4	29.1	32.4	611.9
CHP willow (abandoned land)	-31.8	29.1	32.4	611.9
CHP natural gas	75.2	14.9	85.3	739.5
CHP coal	108.0	20.7	308.1	739.5

The GW impact of gasification willow varies drastically among the considered land scenarios (Table 3) and the assumed land-use references (Table 2). Besides the LCA results being highly sensitive to the displaced energy source, they were also very sensitive to the key modelling assumption of the displaced marginal crop (wheat/barley). Less prominent yet significant, GW impact results were also sensitive to the iLUC emission factors and yield parameters. Considered uncertainties did not dispute the main readings of the presented results, i.e. that substituting coal or natural gas for CHP with gasification willow is overall positive for the environment.

The assessed results show the *transitory* benefits of gasification willow and highlights the need to put disincentives for land-demanding biofuels in the long term, as well as establishing measures or policies on place now to foster low-input, perennial energy cropping systems on abandoned lands with gasification technology and biochar amendment. Alternatively, Denmark can increase its bioenergy potential without undesirable iLUC effects by releasing currently used marginal land through disincentives on land-intensive products.

3.3 Energy return on energy invested

Additionally to the environmental performance indicators shown in Paper **I**, the energy return on investment (EROI) is presented here. Following recommendations to allow a cross-comparison among literature (Murphy et al. 2011; Hall et al. 2011), a cradle to gate EROI ($EROI_{C2G}$) has been calculated as the ratio of *gross* energy output (at the farm gate, based on the lower heating value) to primary energy input (solar excluded). For the primary energy input, the embodied energy in human labour (Stolarski et al. 2014) has also been included, apart from the direct (diesel) energy invested during the farming and transportation operations, the embodied energy in machinery (allocated by the percentage of the lifetime used), fertilizers and pesticides (Ecoinvent 2014). Both marginal scenarios are presented together as they have same inputs and outputs.

Table 4. Summary of the energy performance indicator EROI ratio for low-input willow bioenergy and for several fossil fuels. EROI ratios for fossil fuels from Murphy and Hall 2010.

	Willow (arable)	Willow (marginal)	Natural Gas (2005)	Coal	Imported oil (2007)	Tar sands
$EROI_{C2G}$	37	31	10	80	12	2

The $EROI_{C2G}$ ratios of SRC willow are much higher than the ones of current natural gas or oil (see Table 4), which indicates a good *relative* energy performance of willow bioenergy. These $EROI_{C2G}$ ratios are also remarkably higher than common bioethanol or biodiesel (0.8–6) (Murphy and Hall 2010), but represent the energetic performance of the energy conversion path rather than the cropping system. When embedded energy in human labour and machinery are not accounted for, the $EROI_{C2G}$ ratio of arable willow is 43 which is in the range of other reported $EROI_{C2G}$ ratios (33–44) calculated with higher willow yields (Heller et al. 2003). However, other studies have reported a consistent $EROI_{C2G}$ of SRC willow around 24 (Matthews 2001; Börjesson and Tufvesson 2011; Stolarski et al. 2014). The difference can be partly explained by the fact that they included active drying (instead of a more optimal natural drying (McElroy and Dawson 1986)), fencing and the embodied energy in the transportation.

Interestingly, if EROI is calculated respect to the *net* energy output (per energy delivered), or cradle-to-cradle ($EROI_{C2C}$), it drops to 0.95 and 0.8 for the arable and marginal land cases, respectively, which contrasts with that of wind (20) and photovoltaic (7) power systems (Kubiszewski et al. 2010; Murphy and Hall 2010). This indicates a bad *absolute* life-cycle energy performance of gasification willow for CHP, which is also confirmed by emergy-based approaches (Kamp and Østergård 2011).

3.4 Key learnings

Assessed results suggest that implementing willow gasification for CHP is beneficial in many aspects. Crucial environmental impacts depend on the type of land used for the establishment of the plantation, abandoned lands being the preferred ones. Nevertheless, the stated benefits hold true for the short- and medium-term, or in other words, *as long as* it substitutes natural gas or coal. In a future fossil-free Denmark (The Danish Government 2011), the environmental benefits of gasified willow may be questionable.

Moreover, the additional key learnings from this Chapter are presented below:

- Amending soil with recalcitrant biochar sequestered as much C as it was emitted by fossil sources in the transportation and farming stages. Gasification biochar thus proved to be an important C-sink with great potential. Furthermore, it may also contribute to the sustainability (Glaser 2007) and soil quality (Brandão and Canals 2012) of land use systems.
- Considered GHG emissions from induced LUC effects ($iLUC_{food}$, $iLUC_{feed}$, $dLUC_{aban}$) played a critical role in determining the GW impact of dedicated willow for gasification.
- The developed hybrid $iLUC_{food}$ model showed that transformation of land, more than delayed relaxation or occupation (as increased intensification), dominate the land use impacts in the considered LUC effects. The dominance of transformation (GW and biodiversity impact-wise) over occupation (understood as intensification) was confirmed by the top-down $iLUC_{feed}$ model.
- A more specific framework (like Table 2) is desirable to consistently guide land-use reference choices in land use impact assessments. This is a prerequisite to align LCA results of land-demanding products and land-use systems and to homogenize environmental footprinting results.

4 Time horizon assumptions in LCA of biofuels

Even though different time horizons coexist in LCA, these are not defined in the existing LCA standards (ISO 2006a; ISO 2006b; BSI 2011; ISO 2013). As a result, most practitioners are unaware of the implicit assumptions done about time horizons and the different implications they have on results. A logical analysis is carried out here to distinguish among the different time horizons involved in LCA of biofuels (Paper **II**) and the different long-term impacts that arise from land use and LUC (Paper **III**). The presented time horizon definitions and the derived classification of long-term impacts are seen as prerequisites to frame the analysis of the next study cases (Chapter 5) and the modification proposals of the land use impact assessment methodology of UNEP-SETAC (Chapter 6). However, the presented definitions and categories were the result of a typical iterative process of LCA where the interpretation of results is fed back for the redefinition of the system boundaries and the fine-tuning of the life cycle inventory (LCI) (see Figure 2).

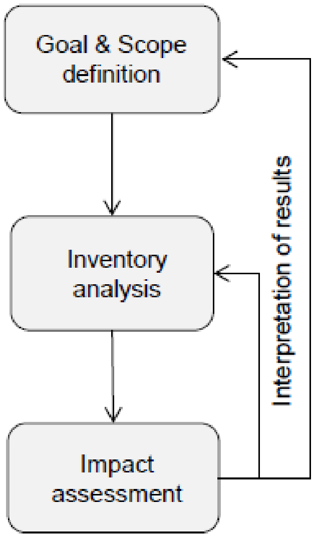

Figure 2. Illustration of the stages and the iterative process in a LCA. Adapted from the standard ISO 2006a.

4.1 Definition of principal time horizons in LCA

The definition and description of the identified time horizons involved in LCA, which affect biofuel assessments too, are presented in Paper **II** and recompiled here below:

- **Amortization period**: Borrowed from financial accounting, the amortization period represents the assumed time horizon over which the assessed activity will take place. Using financial terminology, this time horizon represents the period over which the 'environmental investments' are paid back.
- **Technological time scope**: this time horizon is given by the life-cycle of the assessed product or service and its definition occurs in the goal and scope stage. If a technology (e.g. a smartphone) or the service provided by a technology (e.g. the power delivered by a wind turbine) is assessed, this time horizon refers to the expected lifetime of the technology.
- **Inventory modelling period**: This is the time horizon over which emissions are accounted for and its definition occurs in the inventory analysis stage. In most of the cases, the inventory modelling period ends when the product is disposed of and thus coinciding with the technological time scope. In some cases, significant long-term emissions beyond the end-of-life may be expected, e.g. from landfill (Hauschild et al. 2008) or peat oxidation (Fargione et al. 2008; Valin et al. 2015) and hence they must be included in the LCI to comply with the completeness, relevance and accuracy accounting principles. That is, for such cases an extended inventory modelling period is required (Hauschild et al. 2008; Brandão et al. 2012; Sanchez et al. 2012; Bakas et al. 2015).
- **Impact modelling period**: This is the time horizon used in LCIA methods to calculate the corresponding environmental impacts. E.g. to assess the cumulated GW impact of different GHG emissions in the life-cycle of a product, a 100 year horizon may be used for the global warming potential (GWP) calculation (i.e. the GWP_{100}). Due to the sensitivity of the impact score to the modelling period used in its characterization, this needs to be clearly stated for reporting purposes (ISO 2006b).

4.1.1 Other time horizons in LCA of biofuels

These are other time horizons which are specific to LCA of land-demanding biofuels (and land-based products or land use systems). They are typical of forestry and perennial cropping systems (they coincide in annual crops):

- **Harvesting frequency or single-rotation period**: the time span between harvests, e.g. 3 years for SRC willow.
- **Crop lifetime or full-rotation period**: the timeframe of a crop's full life-cycle, after which the plantation is removed and replanted, e.g. 20 years for SRC willow.

4.2 Long-term impacts from land use and LUC

After the proposed time horizon definitions, three different categories of long-term impacts from land were identified, which are described in Paper **III** and recompiled here below:

- **Pure long-term impacts:** They are impacts related to long-term emissions from LUC (e.g. peat oxidation). Therefore, they start at the beginning of the occupation process (and thus include short-term emissions too) and continue beyond the occupation period. These impacts are *directly* caused by the transformation and occupation of land.
- **Post-occupation impacts:** They relate exclusively to long-term emissions from future land use and/or LUC and start at the end of the occupation period. Rather than being directly caused by occupation, these impacts are *indirect* effects of occupation since they arise when this ends. There are two main options: natural regeneration after abandonment or continued occupation (Milà i Canals et al. 2007).
- **Permanent impacts:** They are irreversible losses of ecosystem services inflicted on land, e.g. sealed land or land with irreversible desertification impacts or the extinction of endemic species. These impacts are *directly* caused by the occupation of land.

4.3 Illustration of time horizon assumptions in a sugarcane ethanol case study

Part of the divergence of LUC emission results (and related biofuel LCA results) come from the different horizons considered and applied in the assessments. To illustrate the variety of such assumptions, a sugarcane ethanol case study is taken. Without showing the final LCA results that these studies derived, the time horizons assumed by each paper to calculate the respective LUC emissions are presented here (see Table 5):

Table 5. Assumed time horizons (in years) by different studies that calculated the life-cycle GHG emissions of sugarcane ethanol production. Values in brackets mean assumptions not explicitly made in the studies. NC stands for not considered.

Studies	Occup. period	Single-rotation	Full-rotation	Technol. time scope	Amortization period	LCI horizon	LCIA horizon
Seabra et al. 2011	(6)	1	6	NC	NC[†]	6	100
Kløverpris & Mueller 2012	(6)[Δ]	1	(6)[Δ]	NC	NC[Δ]	(6)[Δ]	100
Valin et al. 2015	NC	(1)	NC	NC	20-50[*]	20-50[*]	100
LUC_{global} [♣] (Paper **II**)	30	1	6	30	30	30	100
US EPA 2010	30	3	NC[◊]	30	30	20-30[∫]	100

[†] LUC were not considered in this study. [Δ] Stated values refer to hypothetical horizon assumptions, if the same amortization-free method had been applied to a sugarcane ethanol (instead of a corn ethanol) case. [*] 50 years horizon considered in the sensitivity analysis to include long-term emissions from drained peatlands. In these cases, they extended the amortization period (typically 20) accordingly. [♣] Presented and applied in Paper **II** (see Table 6 and Table 8). [◊] The end-of-life management of sugarcane plantations (i.e. removal of stumps, rotary cultivation and replanting of cuttings) was not mentioned in the RFS2 program analysis report. [∫] 20 years horizon considered for SOC losses and 30 years for foregone C-sequestering.

Since calculated LUC emissions are highly sensitive to assumed time horizons (Kløverpris and Mueller 2012), and LCA results are in turn highly sensitive to calculated LUC emissions (Paper **I**), the presented data in Table 5 can explain part of the overall LUC emission variability in literature. It is also a good indicator of the existing uncertainty around critical LUC emissions in LCA of land-demanding biofuels.

4.4 Recommendations about time horizons and long-term impacts in LCA of biofuels

Due to the lack of agreement around critical time horizon assumptions in LCA (see Table 5) and based on the definitions and new category proposals for long-term impacts (sections 4.1 and 4.2), the following recommendations have been derived.

4.4.1 Recommendations for the amortization period

Using again the financial analogy, environmental investments refer to large pulse emissions occurring at the beginning of the life-cycle of a product system and which are fundamental to initiate the project, like LUC of land-demanding products. For such cases, the amortization period refers to the time horizon over which the land is expected to be occupied for the production of the raw material required by the project. That is, the land occupation period sets the grounds for amortization. Therefore, the amortization period in LCA of land-demanding products like biofuels can be defined in two ways:

1. With the full-rotation period –for perennial crops and forestry systems.
2. With the technological time scope, i.e. the technical lifetime of biorefineries, power-plants or biomass-processing plants which use (and demand) the feedstock –for annual and short-lived crops.

In principle, the uncertainty related to long-term land occupation is inherent to annual and short-lived crops. However, energy crops like corn or sugarcane will be replanted as long as they are demanded by existing fermentation plants to produce and supply bioethanol. The technical lifetime of the biomass-demanding plants is a case-dependent rather than user-dependent time horizon and therefore stands as a more objective and relevant choice than arbitrary amortization periods (Searchinger et al. 2008; Plevin et al. 2015; Valin et al. 2015). Moreover, political targets on renewable energy share (European Parliament 2009) and increased energy independency (U.S. Congress 2005) guarantee to a large extent the future demand (and thus the assumed occupation periods) of renewable energy sources like biofuels (Schubert et al. 2008; Bauen et al. 2009; IPCC 2012).

Paradoxically, the full-rotation period of perennial energy crops like willow (20 years) or the technological lifetime of biorefineries and power-plants (around 30 years) is similar or equal to commonly used amortization periods (20-30 years) which are recommended by the European biofuel regulation (European Commission 2009), some environmental footprinting standards (Greenhouse Gas Protocol 2011; ISO 2013) and other life-cycle-based guidelines (The Greenhouse Gas Protocol 2006; Koellner et al. 2013). The suggested amortization criteria are though important for short-lived or annual cropping systems like sugarcane, corn or soybean which are currently used for biofuel production.

4.4.2 Recommendations for the inventory modelling period

In most LCA of land-demanding biofuels, emissions beyond the occupation period are restricted to GHG from drained peatlands, particularly in Malaysia and Indonesia (Valin et al. 2015). For the LCI of these studies, the inventory modelling period should be extended in order to include such long-term emissions. Beyond this exception, it is recommended that the inventory modelling period be restricted to the expected occupation period (i.e. the two stated cases in 4.3.1).

4.4.3 Recommendations for the choice of the impact modelling period and for including pure long-term impacts

The UNEP-SETAC guideline argues in favour of long-term impact modelling periods in land use impact assessments (Koellner et al. 2013). It recommends a 500 years' period in order to cover the long time frame required by land to reach a new steady-state. Taking a long impact modelling period (ideally infinite) is also recommended in Milà i Canals et al. 2007, where the inclusion of land use impacts *after* the actual occupation is defended. Their recommendations were thus based on the inclusion of *permanent* and *post-occupation* impacts, which is here suggested to be differentiated.

On the other hand, if we look at the existing impact categories from land use (Koellner et al. 2013), long-term impacts on land refer to: i) post-occupation impacts, ii) permanent impacts, iii) long-term impacts determined by long-term inventory (rather than impact assessment) methods, e.g. SOC loss representing a long-term reduction on soil quality, or iv) long-term climatic impacts from short- and long-term GHG emissions (Kirschbaum 2003), e.g. the future global temperature change potential (GTP) (Shine et al. 2007) of continued peat oxidation.

The first two types of long-term impacts are dealt with in Sections 4.4.4 and 4.4.5, while the third needs only inventorying. From this new perspective, the only long-term impacts that can be problematic with a conventional 100 year impact assessment horizon relate to time-dependent climatic impacts from GHG emissions beyond the occupation period (i.e. "pure long-term impacts", see Section 4.2). For these special cases, dynamic characterization factors (CF) could be applied and be combined with dynamic GHG inventory models to derive emission-time corrected GW impacts (Levasseur et al. 2010).

4.4.4 Recommendations for including post-occupation impacts

Post-occupation impacts may apply to any land-demanding biofuel and are completely determined by the global (or regional) land market and land-use trends at the end of the occupation period. Nevertheless, determining potential land uses after the occupation period (20-30 years) is highly speculative (Milà i Canals et al. 2007; U.S. Environmental Protection Agency (EPA) 2010; Sanchez et al. 2012). Therefore, post-occupation impacts in land-demanding biofuel LCA are recommended to be included in the sensitivity and uncertainty analysis. From a consequential viewpoint, post-occupation LUC may be seen as 'future indirect effect' of releasing the currently occupied land. Hence, they may be included as 'future iLUC effects' with *negative* sign and immanently high uncertainty.

4.4.5 Recommendations for including permanent impacts

Permanent impacts (e.g. sealed or desertified land) are mainly related to soil quality, biodiversity losses and the deterioration of key ecosystem services from land included in the land use impact assessment methodology of LCA (Koellner et al. 2013). It is suggested that these are dealt with separately and referred to a quantitative threshold, after reaching which the land use acquires a qualitative label of, e.g. "degraded land".

4.5 Key learnings

LCA of biofuels make implicit and explicit assumptions related to different time horizons involved in the methodology, which will undoubtedly affect the reported results (Paper **II**). In the absence of specific definitions in the LCA standards and life-cycle-based guidelines, many practitioners are unaware of their different implications and their influence on results. Agreeing on the definition of the presented time horizons is a prerequisite to perform biofuel LCA in a more transparent, more consistent and a more comparable way. The presented analysis and proposed definitions and categories have led to the following key learnings:

- The expected occupation period is the basis for amortization, and so both periods need to be equal.
- The expected occupation period can be given by the lifetime of the involved technology (the technological time scope), for annual and short-lived crops, or the full-rotation period of perennial or forestry crops.
- It is recommended that: i) *post-occupation impacts* be excluded due to their inherent uncertainty; ii) *permanent impacts* be dealt with qualitatively and separately; and iii) *pure long-term impacts* be restricted to the cases with long-term emissions which go beyond the actual occupation period but are caused by the necessary transformation and/or the intended occupation.
- For the cases where *pure long-term impacts* need to be considered, it is recommended that: i) the inventory modelling period be extended beyond the occupation period; ii) the corresponding climatic impacts be calculated with dynamic inventory methods and dynamic characterisation factors.

5 The effect of modelling approaches in LUC emission accounting

In LUC modelling, combined effects from multiple land-use reference and time horizon assumptions are typically involved, which complicate the analysis and cross-comparison of results. Rather than looking at the sensitivity and uncertainty of economic models to input parameters (Plevin et al. 2015), the present Chapter evens out the involved time horizons (see Table 5) and focuses on the analysis of land-use reference assumptions in different LUC accounting methods (Paper **II**).

To cover a broad range of biofuel types, four energy crops with different plantation life-cycles are selected: oil-palm (25 years), SRC willow (20 years), sugarcane (6 years) and corn (annual). It is assumed that the oil-palm plantation is used for biodiesel production and established on Malaysia at the expense of local rainforest (Wicke et al. 2008), the willow is established on Danish arable land and gasified in a decentral CHP plant ($iLUC_{food}$ of Paper **I**), while the sugarcane and corn are used for ethanol production and are planted on Brazilian rangeland (Lapola et al. 2010) and on US cropland (Plevin et al. 2010), respectively. The LUC emission estimates given by the cited papers, which we name as "*ad hoc* LUC", have been taken as reference value to compare with the LUC emission estimates done with other six LUC accounting methods.

The other LUC accounting methods applied are: a dynamic baseline method (DBM) with two variants (DBM1 and DBM2) (Kløverpris and Mueller 2012; Schmidt et al. 2015); a world-average, top-down LUC factor that includes world-average intensification emissions (LUC_{global}, based on the previous $iLUC_{feed}$ factor of Paper **I**); country-average, crop-specific top-down LUC factors based on the method from the GHG protocol to account for unknown LUC (LUC_{GHGP}) (Greenhouse Gas Protocol 2011), with an attributional and a consequential variant (LUC_{GHGP_A} and LUC_{GHGP_C}, respectively); and the crop-specific, bottom-up (economic) LUC factors of the European Renewable Energy Directive (RED) of 2015 (LUC_{RED15}), which are based on the work of Valin et al. 2015. A detailed description of the methods and the study cases can be found in Paper **II** (see Chapter 11) and in the cited references.

None of these methods describe the different time horizons involved in LCA (Chapter 4) and they generally discern few of them. Due to the different approaches and models taken to estimate LUC emissions, every method assumes different time horizons (see Table 5), but also different land-use references which add up to the biofuels' conundrum: the LUC emission accounting. In this regard, the same criterion to define the crucial occupation period has been applied to all the methods to make them somewhat comparable and to facilitate the analysis of land-use reference effects on results. After the criterion presented in section 4.4.1, the following occupation times for the four study cases (in years) have been assumed: 20 (willow CHP), 30 (sugarcane ethanol), 25 (palm-oil biodiesel) and 30 (corn ethanol). Long-term emissions were not considered in any of the study cases, and so the inventory modelling period is restricted to the stated occupation periods. The commonly applied 100 year horizon is also taken for the impact assessment period, i.e. GWP_{100}.

5.1 LUC emission results with different methods

In order to exclude unnecessary variables that add noise to the already entangled LUC emissions, results are presented per area demanded (see Table 6). The land-use area as functional unit (FU) was thus deliberately chosen to avoid additional yield and energy content assumptions which further complicate the analysis of the effects. Taking the land-use area as FU for LCA of land-demanding biofuels has also been recommended by other studies (Cherubini et al. 2009; Pawelzik et al. 2013).

Table 6. Accounted GHG emissions from LUC with different methods for the four biofuel study cases (in Mg CO_2e ha_{dem}^{-1}). Underlined the LUC estimates closest to the *ad hoc* LUC estimates. DK stands for Denmark, BR for Brazil, MY for Malaysia, US for United States, NA for not accounted. Adapted from Paper **II**.

Energy crops	*Ad hoc* LUC	DBM1†	DBM2$^{\lozenge\Delta}$	$LUC_{global}$$^{\dagger\lozenge\Delta}$	$LUC_{GHGP\_c}$$^{\dagger\lozenge}$	$LUC_{GHGP_a}$$^{\lozenge\int}$	LUC_{RED15}†
Willow (DK)	285$^{\dagger\Delta}$	2	63	<u>206</u>	NA	NA	NA
Sugarcane (BR)	491†	4	95	226	<u>445</u>	205	42
Oil-palm (MY)	629*	5	79	216	<u>451</u>	73	213
Corn (US)	169$^{\dagger f}$	1	95	<u>226</u>	NA	NA	22

† Calculated with a consequential perspective. $^{\lozenge}$ Calculated with a top-down approach. $^{\int}$ Calculated with an attributional perspective. $^{\Delta}$ Includes indirect intensification emissions. ^{f} Back-calculated from reported mean iLUC emission factors. * Calculated as a LUC scenario (native rainforest).

For the Danish willow case, the LUC_{global} gave the closest estimate (28% lower, see Table 6) to the *ad hoc* LUC. These two are directly comparable since they both include intensification effects and the emissions derived from them (only the approach taken differs). The LUC_{global} factor was also the closest one to the *ad hoc* LUC emissions of the US corn case (33% higher, see Table 6). Even though the intensification emissions of LUC_{global} are an intrinsic part of it, this can be subtracted for a better comparison with the back-calculated mean iLUC factor from Plevin et al. 2010, which only include LUC emissions. The 'pure' LUC emissions of LUC_{global} are 165.5 Mg $CO_2e\ ha_{dem}^{-1}$, which are just 2% lower than the *ad hoc* LUC emissions.

For the Brazilian sugarcane and the Malaysian oil-palm, LUC_{GHGP_C} gave the closest estimates (9% and 28% lower, respectively). The *ad hoc* LUC of the Brazilian sugarcane is 311 Mg $CO_2e\ ha_{dem}^{-1}$ if an average LUC (4224 Tg CO2e and 13.6 Mha expansion, Figure 2) is taken (Lapola et al. 2010), rather than the LUC related to sugarcane expansion exclusively on rangeland. In that case, LUC_{GHGP_C} would be 43% higher and LUC_{global} and LUC_{GHGP_A} 27% and 34% lower, respectively (making again LUC_{global} the closest LUC emission estimate). If the crop-specific iLUC factors are taken instead of the crop-group average factors, the LUC_{RED15} gives the closest estimate for the Malaysian oil-palm case (only 10% lower).

5.2 Analysis of land-use reference assumptions in different LUC accounting methods

Here the implicit land-use reference assumptions behind each method are presented on Table 7 and their influence on results (Table 6) briefly discussed. From these two tables and Table 5, it can be inferred that the variability and epistemic uncertainty of LUC emissions in literature can be largely explained from the disagreement around the key time horizon and land-use reference assumptions involved in the different LUC models.

In DBM, LUC emissions from deforestation are accounted for the first year only (from t_0 to t_1), as they are assumed to follow the ongoing regional deforestation trends lagged by one year (Kløverpris and Mueller 2012; Schmidt et al. 2015). Unlike Lapola et al. 2010, Valin et al. 2015, the GHG Protocol method and $iLUC_{food}$, which take several land-use references to calculate *different* LUC on *several types of land*, DBM apply a double land-use reference to calculate LUC on forestland (Paper **II**). All the investigated LUC models but DBM assume that the ecosystems affected by agricultural expansion would have remained in equilibrium in the absence of the studied energy crop. Taking deforestation trends as dynamic land-use baseline, DBM thus include logical circularity in the calculation and, as a result, LUC emissions with DBM are systematically underestimated (see Table 6). In Paper **II**, DBM are thoroughly analysed and the problems with the assumed references discussed.

Table 7. Assumed land-use references for calculating LUC emission estimates from agricultural expansion with the presented LUC accounting methods. REF2 stands for an extensively managed grassland, REF3 for natural regeneration, REF4 for a natural vegetation cover in equilibrium, REF5 for a local arable, semi-intensive land management (see section 3.1), NC for not considered, NA for not applicable (several studies).

Energy crops	*Ad hoc* LUC	DBM	LUC_{global} ǂ	LUC_{GHGP} $^{\Delta}$	LUC_{RED15} ǂ
Willow (Danish cropland)	REF3 + REF4 + REF5	REF4 (t_0-t_1) + REF5 (t_1-t_{end})	REF4 + REF5	NC	NC
Sugarcane (Brazilian rangeland)	REF2 + REF4	REF4 (t_0-t_1) + REF5 (t_1-t_{end})	REF4 + REF5	REF4 + REF5	REF4
Oil-palm (Malaysian rainforest)	REF4	REF4 (t_0-t_1) + REF5 (t_1-t_{end})	REF4 + REF5	REF4 + REF5	REF2 + REF4
Corn (US cropland)	NA	REF4 (t_0-t_1) + REF5 (t_1-t_{end})	REF4 + REF5	NC	REF2 + REF3 + REF4 + REF5

ǂ REF5 refers to the indirect intensification emissions, not LUC emissions of REF5. $^{\Delta}$ The consequential approach considers only the new plantation area (on former cropland, REF5, and forest, REF4). The attributional approach considers the old and the new plantation area, over which the LUC emissions are divided.

The calculated LUC_{RED15} for imported palm-oil considered the conversion of grasslands and forested peatlands (Valin et al. 2015). Including low emissions from grasslands levelled out the 20 years of peat oxidation, which resulted in a similar average to the *ad hoc* LUC emissions from clearing a rainforest. On the other hand, the reported LUC_{RED15} for sugarcane expansion excluded expansion on forests (contrasting with the FAO statistics and the country-specific assessment of Lapola et al. 2010) and included an extraordinary C-sequestration potential. This discrepancy for LUC emissions from sugarcane could be reduced in two manners. First, by selecting, within the assumed land-use reference (REF4), a more realistic type of land converted based on real data (e.g. like the LUC_{GHGP_C} does). Second, by assuming more realistic C-sequestration potential, given that: i) all the plant biomass is processed and burnt in every harvest (Seabra et al. 2011); and that ii) a SOC gain of sugarcane (which requires ploughing for its establishment) with respect to a natural vegetation cover in equilibrium seems implausible. If the assumed

C-sequestration is removed and LUC emissions are reported as non-amortized and per hectare (Valin et al. 2015), it results in 182 Mg CO_2e ha_{dem}^{-1}, which shows greater convergence to the rest of estimates.

5.2.1 Top-down LUC models

These models have the advantage of being simple to calculate, which implies more transparency (Broch et al. 2013) and less parametric and epistemic uncertainty than the complex agro-economic models, because their calculation involves fewer parameters and assumptions. The reporting of LUC emissions is done per occupied or demanded area to support transparency. Because an additional land demand (country-average or global-average) effect is modelled, rather than a world-wide crop-specific indirect effect, key economic assumptions like price elasticity (that determine the magnitude of the final substitution, displacement and replacement effects) are not needed. Hence, top-down LUC models have the potential of providing more solid emission estimates from induced (or unknown) LUC while reducing the uncertainty of case-specific results and the related variability among different studies.

The LUC_{global} factor can be taken as a first proxy for induced land use and LUC emissions globally when an additional hectare is demanded. This can be used as a generic LUC in LCA of land-demanding products with unknown or difficult to estimate LUC. It can also be useful for regulation and/or environmental footprinting purposes, where the inclusion of LUC is necessary (Muñoz et al. 2014) but result homogeneity problematic (Finkbeiner 2014).

The LUC_{GHGP_C} factor has the advantage of being more versatile and precise than the LUC_{global} factor, since it models crop-specific LUC of relevant countries with deforestation. Since it is based on land use cover time series, the critical land-use references and converted types of land need not to be assumed but inferred from the country-specific data. It has thus the potential of providing agreement on these crucial assumptions for LUC modelling.

Even though agro-economic iLUC models include diverse indirect effects (e.g. reduced feed, substitution and yield increases) to simulate the market response to the shock demand and derive the total area expansion, they do not include all the corresponding GHG emissions from these effects. Excluding GHG emissions from second order indirect effects (like a possible 'deforestation rebound effect' in the mid-term from the reduced feed effect in the short-term) can be justified to avoid further uncertainty. Excluding GHG emissions from first order indirect effects (like the increased fertilization that is needed to increase yields) seems unjustified. The magnitude of such emissions in the LUC_{global} factor (2.1 Mg CO_2e ha_{dem}^{-1} yr^{-1}, 29% of LUC_{global} for a 30 year occupation period) suggests that most reported economic iLUC emission factors may be significantly underestimated.

5.3 GHG savings potential of assessed biofuels

To calculate the full life-cycle GHG savings potential of the assessed biofuels, two LUC estimates have been taken: the *ad hoc* LUC and a best-case LUC from abandoned or degraded lands. To the *ad hoc* LUC emissions the SOC changes from cultivation directly inflicted on the occupied land have been also added. These were previously excluded to allow the comparison of LUC emissions calculated with crop-independent top-down LUC factors (i.e. DBM2 and LUC_{global}). Life-cycle GHG emissions from farming, transportation and energy conversion activities have been added. Total GHG emissions are finally compared to the displaced fossil fuel and given as a percentage of the net GHG savings they would bring about if substituted (see Table 8).

Results in Table 8 show that all the assessed biofuels have higher GW impacts than the substituted fossil fuels unless planted on abandoned land. When considering LUC uncertainties, resulting net GHG savings are still not substantial: a maximum GHG reduction of 27% was calculated for the best-case, high-yielding sugarcane ethanol when taking ‘minimum’ LUC emissions from LUC_{global} [1]. When taking the best-estimate for the average LUC of Brazilian expansion instead (311 Mg CO_2e ha_{dem}^{-1}, Lapola et al. 2010), the net GHG saving potential of sugarcane ethanol drops to only 8%. The often claimed environmental benefits of land-demanding biofuels (U.S. Environmental Protection Agency (EPA) 2010; Seabra et al. 2011; Kløverpris and Mueller 2012) are thus once again put into question (Fargione et al. 2008; Searchinger et al. 2008). This statement is done on the basis of other conservative assumptions taken as: i) the applied long amortization periods; ii) the linear, static GWP method applied which underestimates the total GW impact potential of biofuels (O’Hare et al. 2009; Levasseur et al. 2010); iii)

[1] LUC emissions from DBM were rejected for this uncertainty analysis due to the explained problems and discussed flaws in Paper **II.**

the exclusion of potential long-term emissions from peat oxidation; iv) the crediting of substituted co-products; v) the high yield estimates assumed; and vi) the LUC uncertainties considered. On the other hand, ensuring environmentally beneficial energy cropping on marginal abandoned lands entails other challenges related to their economic volatility, which are more thoroughly discussed in Papers **I-II.**

Table 8. GHG emissions of different biofuel scenarios. BF stands for biofuel and include farming, transportation and energy conversion activities. FF stands for fossil fuel. Adapted from Paper **II.**

Biofuel scenarios	**Total LUC emissions** [a]	**GHG emission factors** ($g\ CO_2e\ MJ^{-1}$)				**Net GHG savings**
	($Mg\ CO_2e\ ha_{dem}^{-1}$)	**LUC** [a]	**BF**	**BF + LUC**	**FF** [b]	(% of FF)
CHP gasification willow (DK)					75.2	
Cropland	282.6 [a]	87.9	-0.8	87.1		-16
Abandoned cropland	84.6 [a,c]	33.1	0.6 [d]	33.7		55
Sugarcane ethanol (BR)					92.5	
Rangeland	565.6 [a]	117.6	20.5 [e]	138.1		-50
Abandoned pasture	107.7 [a,c]	22.4	20.5 [e]	42.9		53
Palm-oil biodiesel (MY)					88	
Rainforest	702.2 [a]	145.2	26.6 [ef]	171.8		-95
Degraded grassland	82.9 [a]	17.8	26.6 [ef]	44.4		50
Corn ethanol (US)					92.5	
Cropland	169.0	62.0	65.0 [e]	127.0		-37
Abandoned cropland	48.0 [c]	17.6	65.0 [e]	82.6		11

[a] Including dLUC (SOC changes) from cultivation. [b] Values correspond to natural gas (in a decentralized CHP), gasoline (for bioethanol) and diesel (for biodiesel). [c] CO_2 emissions as foregone C sequestration. [d] Lower yield was considered for the abandoned land. [e] Substitution effects of co-products were included. [f] Oil-palm trees were assumed to be removed at the end of their life-cycle (but temporary C-sequestration was considered).

5.4 Key learnings

The variability and epistemic uncertainty of LUC emissions in literature can be largely explained from the disagreement around the key time horizon and land-use reference assumptions involved in the different LUC models (Tables 5, 6 and 7). The definition proposals and recommendations from Chapter 4 can help reducing the variability related to time horizons in LCA of biofuels. As a second step to reduce further the corresponding variability and uncertainty, a generic framework (like the one shown in Table 1) can set the grounds to increase the transparency and consistency related to the application of land-use reference assumptions in LCA and LUC modelling. Furthermore, converted land types could be determined with more precision through the consequential variant of the GHG Protocol method. The accuracy of the method to estimate converted land types can be improved with higher resolution of land cover data from the regions/states affected by deforestation for key and diverse countries like Brazil.

Additionally, the following key learnings have been derived:

- Dynamic baseline methods present several methodological problems related to their basic land-use reference assumptions and their application is rejected for LUC emission accounting in biofuel LCA.
- Land-use area is a more appropriate FU for LCA of land-demanding biofuels and for reporting LUC emissions, since it increases reporting transparency and allows for comparison with bio- and fossil fuels.
- Top-down LUC emission factors can be a complementary way to estimate LUC bypassing systemic uncertainties inherent to economic iLUC models.
- Due to higher transparency and fewer parameters and assumptions involved in the top-down models, proposed LUC emission factors reduce result uncertainty and can narrow down the range of results in future literature. They may thus provide an alternative for controversial LUC emission regulation and more accurate C-footprinting.
- The LUC_{global} factor can be considered as a first proxy for induced GHG emissions from land use and LUC when an additional hectare is demanded.
- Land-demanding biofuels have a higher GW impact than the counterpart fossil fuels they substitute unless planted on abandoned lands.

6 Modification proposals for the UNEP-SETAC land use impact assessment framework of LCA

Previous learnings are assembled here to build a proposal of methodological changes to the existing UNEP-SETAC land use impact assessment framework. To the time horizon and long-term impact category definitions (Chapter 4), another classification is added here which structures the new methodology proposal: environmental flows and stocks (Paper **III**). Likewise, different land-use reference frameworks adapted to different LCA modelling approaches are suggested. Value choices need to be made during different steps of LCA, but they need to form a coherent whole (Hofstetter 1998). Consequently, value-consistent land-use reference frameworks are proposed to be used depending on the intended application: a consequential LCA (CLCA), an attributional LCA (ALCA) and a proposal of a hybrid LCA (HLCA).

The novelty of this new methodology regards the change of the overall approach, since it *focuses on land use impacts during the occupation process*. This is enabled through the distinction of different long-term impacts, which are dealt with separately (see Chapter 4), and through the classification of ecosystem services provided by land, which are treated like environmental *flows* or *stocks* (see Table 9). This last discrimination is based on the intrinsic dynamics of natural forces which regenerate the materials that are crucial to human life at different paces, e.g. water recharge cycles vs. C-stock generation timescales.

Table 9. Suggested classification of existing midpoint level indicators in the UNEP-SETAC framework for land use impact assessments in LCA. NPP stands for net primary production, HANPP for human appropriation of NPP. Taken from Paper **III**.

	Midpoint level indicators	Related ecosystem quality or service
Environmental stocks	SOC	Soil quality
	Biodiversity	Ecosystem quality and resilience
	C stocks	Regional and global climate regulation
Environmental flows	NPP, HANPP, SOC changes	Biotic productivity
	Water recharge and purification	Freshwater provisioning and quality
	Soil erosion	Biotic productivity

Concerning long-term impacts on land can be then conceived as *permanent* impacts that irreversibly deplete key environmental stocks (and which affect the related ecosystem quality/services), or those that irreversibly impair the land by reducing or destroying its ability to provide key ecosystem services that behave like flows. Moreover, the distinction between environmental flows and stocks opens the possibility of advancing LCA-based *relative* sustainability assessments to much needed *absolute* sustainability assessments. LCA-based land use impact assessments could be linked to the existing Planetary Boundaries framework (Rockström et al. 2009) with the normalization method proposed by Bjørn and Hauschild 2015. Planetary and regional boundaries specific to land can then be defined as minimum environmental stock (e.g. minimum forest cover area) or maximum environmental flow (e.g. maximum N and P flows to water basins) thresholds per land-use area, which cannot be surpassed to keep the Earth's ecosystems within a "safe operating space" (Steffen et al. 2015). For carrying out such assessments, selecting an area-based FU would be thus more convenient. Several normalization references for land use impact categories would result with this new approach (see Section 7.1), but these have not been calculated.

6.1 Land use impact calculation

The new approach forced to slightly modify the existing land use impact calculation methodology. Since defining the occupation period is inevitable in land use impact assessments (see Chapter 4), a single type of land flow (area transformed/occupied) results for the LCI. This also allowed the proposal of a generic formulation for land use impact assessments (see supplementary material 4 in Paper **III**). The CF for land use impact assessments are thus modified and generically redefined as the induced change in a certain ecosystem quality (ΔQ) by the assessed land use relative to a land-use reference, which is specific to the type of land transformed/occupied. The land-use reference is fully determined in conjunction with the selection of the LCA modelling approach (see Table 10) and the type of land use impact (transformation/occupation).

OI are split into two categories: delayed relaxation impacts (DRI), when occupation postpones natural regeneration (i.e. on abandoned lands), and simple OI (see Figure 3) for the rest of land occupations. For DRI, the land-use reference is *dynamic* while it is *static* for OI and TI. The total of land use impacts is thus the sum of the individual impacts involved, which are given by the relevant CF (i.e. CF_{DR}, CF_{occup}, CF_{transf}) and the area of land transformed/occupied.

Even though every land intervention affects both environmental flows and stocks, transformation processes are expected to dominate land use impacts affecting environmental stocks with this methodology. Similarly, despite long occupation processes affecting environmental stocks, e.g. SOC or biodiversity via permanent habitat fragmentation (Swift and Hannon 2010), they are expected to dominate land use impacts affecting environmental flows, e.g. soil erosion or other than the ones covered by the UNEP-SETAC framework (e.g. N flows into water basins), because they accumulate over the whole occupation period.

6.1.1 The FU and the calculation of land use impacts on environmental stocks and flows

Typically, an LCA focuses on an individual *product functionality* and practitioners choose the FU accordingly, e.g. 1 MJ in biofuel LCA. In such LCA, impacts are *amortized* over the total (life-cycle) production, implying that land use *impacts on environmental stocks are amortized* (e.g. LUC emissions, see Table 8). Hence, amortization indirectly transforms land use impacts on stocks (e.g. total C losses after the occupation) into flow-equivalent impacts, e.g. in Mg C ha_{occup}^{-1} yr^{-1} or g CO_2e MJ^{-1} (yr^{-1}). On the contrary, if a practitioner chooses a land area as FU instead (e.g. 1 ha), *land use impacts on environmental flows are aggregated* (see Figure 3). Hence, aggregation transforms land use impacts on flows (e.g. the annual loss of groundwater recharge after the occupation) into stock-equivalent impacts (i.e. a total loss of groundwater, in mm H_2O ha_{occup}^{-1}).

On one hand, product-based FU entails allocation dilemmas for land use systems with co-products. On the other, area-based FU implies land-demanding products being indirectly compared through the respective land use systems. If co-products are involved, allocation is avoided but direct comparison of products beyond the individual LCA (i.e. footprinting and eco-labelling) is not possible.

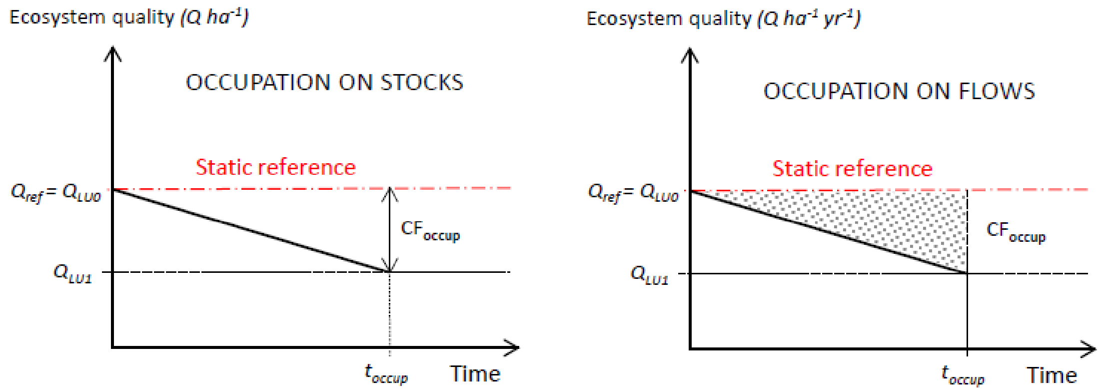

Figure 3. Illustration figure for the calculation of generic OI on environmental stocks (left) and on environmental flows (right) in land use impact assessments. OI on the right (e.g. a groundwater recharge potential loss) is independent from the impact on the stock presented on the left (e.g. a SOC loss). Taken from Paper **III.**

6.2 The hybrid LCA proposal

The new calculation methodology can be adapted to existing modelling approaches (CLCA and ALCA), applying the corresponding land-use reference frameworks and including additional CF (CF_{BAU} or CF_{DR} for business-as-usual (BAU) or DR impacts, respectively, see Paper **III**). Besides these two, a third modelling approach was suggested (HLCA), with the intention of including key iLUC effects and enabling absolute impact assessments at the same time. For this, the use substitution is excluded (see Chapter 7) and iLUC are incorporated with top-down LUC models (see Section 5.2.1). Considering DRI may introduce a substitution element, thus their inclusion in the framework is considered *interim*. If DRI are excluded, the precedent vegetation cover in equilibrium would be the only land-use reference for HLCA (see Table 10).

Table 10. Land-use reference framework for land-use impact assessments with HLCA. All the land-use reference frameworks are shown in Paper **III**. Adapted from Paper **III.**

Land status	Used		Unused	
Induced LUC	Yes §		No	
Precedent land-use	Arable land, grassland, built-up land, managed forest		Marginal abandoned land †	Natural and semi-natural ecosystems
Land-use impact	Occupation	Transformation	Delayed relaxation◊	Transformation
Land-use reference	Static reference (vegetation cover of precedent land use in steady-state)		Dynamic reference (natural regeneration)	Natural vegetation cover in steady-state

§ Including iLUC effects compulsory, if the studied land use system or the land-based product creates an *additional* demand on land. † Including degraded land and land at risk of abandonment. ◊ Interim. DRI may include a substitution effect which is incompatible with the assessment of absolute impacts.

6.2.1 Dynamic reference in attributional LCA

The inconveniences related to natural regeneration as a single land-use reference for ALCA studies are thoroughly discussed in Paper **III**. Clearly, its appropriateness depends on the (not agreed) definition of ALCA and its intended application(s) (Brander et al. 2009; Plevin et al. 2014). Depending on the definitions and applications of ALCA and CLCA (possibly agreed and standardized in the future), the existence and application of HLCA could be reconsidered. The proposed HLCA borrows from the CLCA the inclusion of induced LUC (although not sharing the methodology). Its overall approach may be though closer to ALCA and in line with the intention behind the dynamic reference proposal (Soimakallio et al. 2015; Soimakallio et al. 2016), which arguably intends to represent land use impacts with an ecocentric perspective. HLCA is thus an alternative way of satisfying the aim of the dynamic reference proposal for ALCA (see Chapter 7). In this respect, the presented HLCA framework should be seen as a first attempt to reconcile top-down with bottom-up modelling approaches for the assessment of land-intensive product systems like dedicated bioenergy (Creutzig et al. 2012).

6.3 Improvements to the UNEP-SETAC framework.

Several problematic assumptions needed in the previous UNEP-SETAC methodology are avoided with the suggested changes. Specifically, assuming transformation as instantaneous is no longer an issue and the ecosystem quality of the assessed land use can vary during the occupation process. Clear interdependencies between climatic and soil quality impacts (they both directly depend on the induced SOC loss on land) can be now considered with unit consistency. The mentioned land-use reference frameworks can also allow value-consistent land use impact assessments for different purposes and applications. Last but not least, ecosystems' tipping points can be equally incorporated through the HLCA framework with the suggested normalization method (Bjørn and Hauschild 2015) and an area-based FU (see discussion in Chapter 7).

6.4 Key learnings

With the suggested methodology and frameworks, the inputs of the LCA practitioner to calculate land use impacts are restricted to:

- The *object* of the study, i.e. the crop, land use system or land-demanding product. This determines the expected occupation period.
- The *objective* of the study and its intended application, i.e. the modelling LCA approach adopted. This determines the FU of the study.
- The *location and land type* required by the respective land use system. This determines, together with the land use system, the need to consider permanent impacts and long-term inventory and impact assessment horizons.

Natural regeneration or dynamic land-use references are suggested for the assessment of occupation impacts on abandoned lands and be considered as delayed relaxation. Their use may be more appropriate in CLCA, since it arguably introduces a substitution element (i.e. a ‘negative emission’). A static reference representing a steady-state vegetation cover is recommended to assess occupation and transformation impacts in all the frameworks. With this methodology, additional impacts relative to a business-as-usual reference (CF_{BAU} for CLCA) or relative to a dynamic reference (CF_{DR}, as in the current proposal for ALCA) can be included without losing consistency.

The hybrid framework has been presented as an alternative LCA modelling approach to ‘pure’ attributional and consequential studies which can perform both *relative* and *absolute* sustainability assessments. This is done by including important market-mediated LUC effects through top-down average LUC factors (for the former) and by excluding substitution and incorporating planetary and regional boundaries through area-based carrying capacity thresholds as normalization references (for the latter). Notwithstanding, the articulation of absolute impact assessments with area-based FU has a high price: excluding substitution would imply excluding allocation, thus excluding the possibility of product footprinting in every land use system with multiple outputs (i.e. with co-products). The methodology allows anyway carrying out relative LCA with product-based FU and common allocation rules.

7 Discussion

Presented results and suggested recommendations in Chapters 3-6 have been already briefly discussed and they are more thoroughly analysed in Papers **I-III**. The author has deemed necessary to focus the following discussion on the limitations and challenges of LCA to perform *absolute* land use impact assessments. Given the lack of consensus around the definition of ALCA and the appropriate land-use reference(s) for it, the HLCA was proposed with the aim of articulating such assessments. However, this entails several challenges, since the whole LCA methodology has been conceived as a *relative* sustainability assessment tool. Absolute assessments will be nevertheless crucial for future decision-support on land management and other environmental policies. Since this Thesis follows the post-normal science paradigm, they are henceforth addressed.

7.1 Relative vs. absolute impact assessments

All LCA studies perform substitution of 'avoided' environmental impacts (although it has been put into question for ALCA by Brander and Wylie 2011), thus implicitly giving for granted credited impacts of anthropogenic origin. That is, existing LCA modelling approaches take an *anthropocentric* perspective. On the contrary, the HLCA framework tries to incorporate the planetary boundaries framework (Rockström et al. 2009) to enable *absolute* sustainability assessments with the mentioned method (Bjørn and Hauschild 2015), but additionally excluding substitution. That is, the HLCA framework takes an *ecocentric* perspective. It is here suggested that *substitution must be excluded in order to perform absolute land use impact assessments with LCA*. The main reason being that the planetary boundaries framework takes the natural carrying capacity of ecosystems as reference point, while common LCA take the current consumption and emission level of studied societies as reference point. References need to be aligned though in such assessments.

Let us think of classical mechanics for a moment. To determine the physical properties of a solid material (e.g. velocity, momentum, etc.), any reference system may be taken. The resulting properties (e.g. $\boldsymbol{v'_1}$) are expressed in relative terms, i.e. relative to the reference system $\mathfrak{R}_1$ (see Figure 4). In order to determine the absolute properties of the solid ($\boldsymbol{v_1}$), the properties of the reference system ($\boldsymbol{v_0}$) must be added ($\boldsymbol{v_1} = \boldsymbol{v'_1} + \boldsymbol{v_0}$).

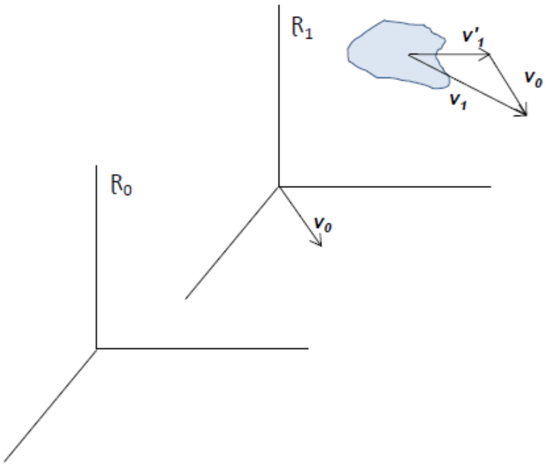

Figure 4. Graphical illustration of relative and absolute reference systems (or frame of reference) for physical property calculations in classic mechanics. R_0 is the absolute reference system, R_1 is the relative reference system and $\boldsymbol{v_i}$ represent the velocity vectors.

By analogy, an absolute reference system $\mathrm{R}_{\boldsymbol{0}}$ must be taken to carry out absolute environmental impact assessments. Since $\boldsymbol{v_0}$ must be zero for a $\mathrm{R}_{\boldsymbol{0}}$, it corresponds to the natural *equilibrium* of ecosystems. If the emissions or ecosystem services/quality (denoted *e*) are measured relative to a social reference system $\mathrm{R}_{\boldsymbol{1}}$ (i.e. as e'_1) characterised by the business-as-usual emissions e_0, the credited emissions from substitution need to be added to get the absolute emissions ($e_1 = e'_1 + e_0$). Since LCI typically present emission data relative to reference ecosystems (e_1), it implies that absolute impact assessments should disregard e_0. That is, absolute assessments should not credit substitution emissions as it converts absolute emission impacts to relative impacts ($e'_1 = e_1 - e_0$), like common LCA do (because they are commonly relative assessments).

In Table 11 it is shown the life-cycle results of two different indicators previously presented, which illustrate numerically what it has been argued theoretically. The presented relative environmental indicators show lower impact scores than the relative reference because of the credited emissions. Also the $EROI_{C2G}$ indicator shows better results than the relative reference. However, the reading of the same results radically changes when an absolute perspective is taken. Gasification willow goes from being almost C-neutral to being highly C-intensive. Low-input SRC willow goes from a very good energetic performance (when compared with natural gas), to a poor performance from an absolute (cradle-to-cradle) perspective (as the whole bioenergy system requires more energy than it delivers, like most fossil fuels).

Table 11. Selected performance indicators for SRC willow cultivated on Danish arable land and used for CHP with gasification and biochar amendment.

	Relative	Absolute	Unit	Sources
GW_{100}	0.8	87.1	g CO_2e MJ^{-1}	Tables 3 and 8
Reference	Natural gas CHP	Earth's atmosphere		
Reference value	75.2[§]	0	g CO_2e MJ^{-1}	Table 8
EROI	37	0.95	[-]	Table 4
Reference	Natural gas (C2G)	Net energy neutrality		
Reference value	10	1	[-]	Table 4

[§] Emissions calculated per MJ of fossil energy content. The bioenergy-equivalent GHG emissions are 86.2 g CO_2e MJ^{-1} (the credited amount), corrected with the energy content of willow woodchips.

By incorporating top-down LUC factors and not allowing for substitution, the HLCA framework intends to facilitate both *relative and absolute* impact assessments, since relative assessments can always be carried out with any agreed, common reference system. Combining the HLCA framework with an area-based FU, the assessment and comparison of land use systems (i.e. taken as *product systems*), rather than the assessment and comparison of land-using *products*, is articulated. So long the land use systems have one product as single output (e.g. SRC willow for bioenergy), the HLCA framework can also be used to assess and compare equivalent products, i.e. footprinting (see Section 6.1.1).

7.2 Bringing the UNEP-SETAC framework beyond: envisioning integral and absolute sustainability assessments of land use impacts

Based on the discrimination proposal of environmental flows and stocks, and given the proximity and transgression of tipping points in several planetary boundaries (Steffen et al. 2015), it can be argued that the sustainability of land generally implies that:

- harvests do not exceed the regeneration rates of renewable resource flows;
- extraction rates do not exceed the substitution rate of non-renewable resource stocks by equivalent human and renewable natural capital (Perman et al. 2011);
- emission rates do not exceed the assimilation or carrying capacity of ecosystems (Goodland 1995; Sayre 2008).

These translate into ensuring a limit to the maximum extraction/emission rates of key environmental *flows* as:

- Freshwater extraction (Foley et al. 2005; Bjørn et al. 2014)
- Biotic production extraction or human appropriation of NPP (Vitousek 1997; Haberl et al. 2010; Running 2012; Smith et al. 2012)
- CO_2 and other GHG emissions (Pachauri et al. 2014; Steffen et al. 2015)
- N and P emissions to oceans and freshwater basins (Steffen et al. 2015)

As well as a limit to the absolute depletion of key environmental *stocks* as:

- Biodiversity, both functional and genetic (Barnosky et al. 2012; Steffen et al. 2015)
- Biologically sequestered carbon (Righelato and Spracklen 2007; Schulze et al. 2012; Smith and Torn 2013).
- Fertile and natural land (Steffen et al. 2015).

The last “land-system change” boundary has some clear interactions with regional climate (Steffen et al. 2015) and biodiversity too, as it is affected by the fragmentation of land (Swift and Hannon 2010) and the clearing of natural forests (Noss et al. 2012). The C-stock and biodiversity thresholds may thus be integrated and more easily represented (as first proxy) by a percentage of minimum forested land cover, regionally and globally.

Linking environmental impacts to ecosystem carrying capacity thresholds is relatively straight-forward if an area-based FU is taken. The area-based entitlement shall be according to the relevant geographical scope of the boundary or impact category, e.g. global area for planetary boundary thresholds like radiative forcing (for the GW impact category), water-basin area for freshwater extraction and P input thresholds (for those related to water use and eutrophication impact categories, respectively). Note that the normalization method to introduce ecosystem equilibrium thresholds into LCA (Bjørn and Hauschild 2015) can only link those impact scores for which relevant planetary and regional boundaries have been identified, i.e. the extraction of freshwater and emission of N, P, CO_2 and other GHG (for the environmental flows) and the land area transformation/occupation thresholds (as a first proxy for the relevant environmental stocks).

Notwithstanding, it must be highlighted that *out of the fifteen* proposed control variables to guide human development within the biophysical limits of the Earth (Steffen et al. 2015), *seven are directly and exclusively associated to land use.* Four out of these seven control variables are metrics of environmental stocks[2] while three are metrics of environmental flows[3]. Two additional control variables (radiative forcing and atmospheric CO_2 concentration) are both a metric of an environmental flow from land (biotic CO_2 emissions) and its respective stock (biotic C pools), but these are neither exclusive to land use (e.g. fossil CO_2 emissions) nor exclusive to CO_2 (other GHG[4] contribute to the total radiative forcing that induce global warming). From the remaining six control variables, three of them are indirectly or partially related to land use and LUC[5]: ocean acidification (since the carbonate ion concentration increases with the total CO_2 concentration in the atmosphere) and freshwater use (since the withdrawals from river basins, regionally and globally, is highly affected by irrigation systems). Therefore, *twelve out of fifteen control variables are directly or indirectly related to land use and LUC*, which shows the importance of advancing the UNEP-SETAC framework towards absolute land use impact assessments.

7.2.1 Sustainable land management criteria

In the light of the planetary boundaries evidence and keeping in mind the environmental flow and stock distinction of Chapter 6, sustainable land management criteria shall generically imply both:

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[2] Species extinction rate, biodiversity intactness index and remaining forest cover (at two spatial levels: globally and regionally).

[3] P flows to aquatic systems (globally and regionally) and global N fixation (mainly industrial, for fertilizer production, but also intentional biological, such as the cultivation of legumes). They are key environmental flows that threaten marine and freshwater ecosystems with eutrophication.

[4] Many GHG emissions (mainly CH_4 and N_2O) come though from land use and LUC, namely enteric and anaerobic fermentation processes, fertilizer and manure application, agricultural residues, wild fires and rice cultivation (Smith et al. 2014). Black carbon aerosols (from bioenergy combustion) have been also found to be an important disturbing climate force capable of changing tropical monsoon patterns (Feng et al. 2013).

[5] Around 12% of global CO_2 emissions that end up absorbed by oceans come from terrestrial ecosystems (Smith et al. 2014) and a large part of the water withdrawals from lakes and river basins is for agricultural purposes, i.e. irrigation (Steffen et al. 2015).

- *Minimizing transformation* processes to preserve a minimum of key environmental stocks and
- *Optimizing occupation* processes (including delayed relaxation) to minimize the emissions/extraction of key environmental flows and maximize existing key environmental stocks.

The first implies a *public landscape management* that ensures minimum environmental stocks that go beyond the individual land plot. The responsibility of this management to set the appropriate regional boundaries (and to monitor and control that they are respected), ultimately lies on the relevant public entities, although its implementation may depend on private forest management agents (Cintas et al. 2015). The second implies a *private land management* that maximizes agricultural production *subject to* the absolute thresholds of key environmental flows (i.e. water extraction and N and P emissions). A full optimization of private land management shall involve the maximization of key environmental stocks (i.e. C stocks and biodiversity) too, e.g. through the combination of different land sharing (agro-ecology, agro-forestry or permaculture practices) and land sparing strategies (Phalan et al. 2011).

Both environmental and ecological economic schools arguably agree on *the level and extent* to which human capital can substitute key natural capital at a planetary scale (Costanza et al. 1997; Huesemann 2004; Perman et al. 2011). Hence, a new generic "weak sustainability" criterion[6] for global land use impact assessments is suggested, which can be defined qualitatively as a *stock-neutral land management* (where e.g. SOC or biodiversity do not decrease) *which ensures that key environmental flow extraction/emission thresholds are not surpassed*. On the other hand, a new generic "strong sustainability" criterion is proposed as the land management that increases the ecosystem's resilience, making it stronger against future disruptions or extreme events (Holling 2001) and compensating for the generalized loss of key stocks around the globe. This can be qualitatively defined as a *stock-increasing land management with decreasing key environmental flow extractions/emissions*.

[6] Taking the environmental "strong sustainability" definition implies maintaining the natural capital constant, while the "weak sustainability" definition implies maintaining the total capital (natural and human) constant (Perman et al. 2011). This is based on how the different schools see the fundamental question of capital *substitutability*, i.e. *the form* in which human-made assets can substitute the environmental services provided by nature.

8 Conclusions and recommendations

Different sources of epistemic uncertainties in LCA of land-demanding biofuels were discovered with different methods. After a holistic analysis, it has been found that the sources of epistemic uncertainties in land use impact assessments (including LCA of dedicated bioenergy) regard mainly:

- The lack of agreement around the applied land-use references.
- The lack of definitions for the different time horizons involved in LCA.
- The nature of the applied iLUC emission factors, which require complex and black-box agro-economic models with many parameters and assumptions, the uncertainty of which might even be difficult to assess.
- The lack of agreed definitions and applications for the different LCA modelling approaches which enable different sets of assumptions.

To reduce the epistemic uncertainty derived from the first and the last sources, value-consistent land-use reference frameworks have been proposed (Chapters 3 and 6). For the second, time horizons and long-term impacts have been differentiated and defined (Chapter 4). For the third, less precise but more simple and transparent top-down LUC models have been proposed instead of economic iLUC factors (Chapter 5). Economic iLUC models might be still relevant for policy analysis purposes with CLCA.

Moreover, several important learnings were achieved with this Thesis. The most relevant research output is presented first and the different recommendations after:

- Studied gasification technology with soil amendment of the biochar-bioash residue showed promising results with significant C-sink potential.

- Soil amendment with a biochar-bioash mix likely has further environmental potential benefits on agricultural soil's quality and on P-K resource savings. Final effects depend on specific soil types and thus generalized benefits cannot be confirmed based on current knowledge.

- GHG emissions from demanding additional land (modelled as transformation, delayed relaxation and intensification impacts), play a crucial role in determining the GW impact of land-demanding bioenergy systems.

- Among the induced indirect land use and LUC effects, transformation impacts dominated more than occupation (modelled as additional intensification) or delayed relaxation impacts.

- The occupation period of land is the basis for amortization of LUC impacts
- The expected occupation period is more objectively given by the lifetime of the involved technology (for annual and short-lived crops) or by the full-rotation period (for perennial crops).
- Dynamic baseline methods present several methodological problems related to their basic land-use reference assumptions and their application is rejected for LUC emission accounting in LCA of land-demanding biofuels.
- The land use area is a more appropriate functional unit for reporting LUC emissions as it increases transparency. This also allows a better comparison of LUC emissions since it excludes uncertain and critical yield estimates.
- An area-based functional unit is also more convenient to perform absolute impact assessments with the HLCA framework, although it impedes product-based emission profiles (i.e. environmental footprinting) in land use systems with multiple outputs (i.e. with co-products).
- Top-down LUC models involve fewer parameters and assumptions than economic iLUC models and thus have less epistemic and parametric uncertainties.
- Top-down LUC emission factors can potentially narrow down the range of future LUC results, and thus may provide a better alternative to economic iLUC factors for regulation and C-footprinting.

8.1 Recommendations for land use impact assessments in LCA

After the proposed methodological modifications, the choice of the LCA practitioners is restricted to:

- The *object* of the study, i.e. the crop or land use system.
- The *objective* of the study and its intended application, i.e. the modelling LCA approach adopted. This determines the FU of the study.
- The *location and land type* required by the respective land use system. They determine, together with the land use system, the need for including permanent impacts and long-term LCI and LCIA horizons.

Similarly, it is recommended that different long-term impacts on land are differentiated and be addressed according to the application and/or specific case. For land-demanding biofuels, it is recommended that:

- *Post-occupation impacts* be excluded for their inherently high uncertainty (future '*–iLUC*' effects).
- *Permanent impacts* be dealt with qualitatively and separately.
- *Pure long-term impacts* be restricted to peat oxidation cases (or other exceptional cases with 'pure' long-term emissions).

8.2 Recommendations for policy-makers on land-demanding bioenergy

- Energy cropping should be incentivized only on marginal abandoned lands of the same country or political union, or on degraded land with third-party accountant certification for imported feedstock.
- To satisfy the demand of energy applications with no (or difficult) alternative, the implementation of quotas could be considered in the future for energy cropping on productive land (and preferably approximated by the abandoned land estimates).
- Existing incentives for energy cropping on arable or productive land should be progressively removed to avoid further iLUC effects.
- Biofuels from land-demanding feedstock should be in any case seen as an acceptable alternative to fossil fuels as long as it is part of a transition strategy towards fossil-free or low-carbon societies.

8.2.1 Recommendations for Danish policy-makers

- Environmentally friendly bioenergy cropping can only come at the expense of marginal grasslands (permanent extensive pasturelands) currently used for cattle grazing.
- If these marginal grasslands are not released from their current demand, their occupation for energy cropping will trigger indirect effects with adverse impacts on global biodiversity and vulnerable population (likely sparking social conflicts).

9 Future perspectives

The feasibility and limitations of the HLCA framework should be further explored and the convenience of its approach with respect to a ‘pure’ ALCA approach with a dynamic reference further studied. This would allow to clarify the trade-offs of each approach and either discard one of them (if one is proven to be superior for the same intended aim) or delimitate the potential (different) applications of each of them.

Besides testing and exploring the new HLCA framework, two general areas for future research have been identified. The first regards the quantification of uncertainties of top-down models, more specifically:

- The quantification of the parametric uncertainty ranges of top-down LUC models with statistical methods (e.g. Monte Carlo simulations).
- The inclusion of albedo effects in top-down LUC models to compute the net GWP of LUC, as well as the incorporation of other impacts besides GW, e.g. biodiversity, change in water cycles affecting local communities (reduced rainfall and groundwater infiltration), etc.

The second area for future research regards the future development of the UNEP-SETAC framework, more specifically:

- The development of carrying capacity based normalization references for absolute land use impact assessments for all the existing midpoint level indicators of the UNEP-SETAC framework.
- For the impact categories without corresponding planetary or regional boundaries (e.g. freshwater purification or soil quality), the potential natural vegetation *in dynamic equilibrium* could be taken as normalization reference. For the case of biotic productivity, the total biotic stocks (rather than the biotic flows, i.e. NPP) of the assessed land use system could be taken as normalization reference.
- The application of the new methodology and frameworks to selected study cases.
- The development/exploration of one or two endpoint level indicator candidates, e.g. total C stocks and functional/genetic biodiversity. The concept of *hemeroby* could be used as inspiration (Brentrup et al. 2002; Fehrenbach et al. 2015) for deriving normalization references for them.

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